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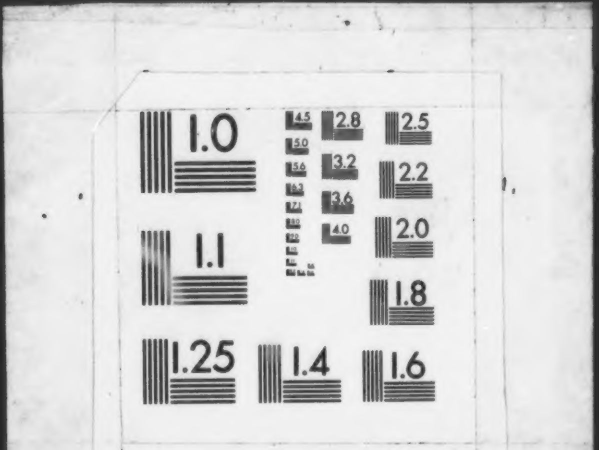
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This volume represents EPA's effort to take stock of scientific advances in research pertaining to the coastal environment since the inception of the Agency and to gauge what progress has been made and what remains to be accomplished. The essays in this volume present a range of perspectives on the subject, from the vantage points of the scientific and technical disciplines which have been carrying out relevant research. The points of view represented are varied and sometimes conflicting. The authors have speculated about emerging problems and research needs. Such attempts require extrapolation based upon informed scientific judgment.

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Impact of Man on the Coastal Environment

Technical Editor

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FOREWORD

The formation of the U.S. Environmental Protection Agency in 1970 ushered in the first decade of environmental awareness as a total national phenomenon. It was a decade punctuated by major Congressional mandates to restore the nation's waters, to reduce air pollution, and to find a comprehensive approach to other environmental problems—those associated with pesticide use, hazardous waste disposal and toxic substances. It was a decade underscored by the demand for new technology and better science to answer environmental questions and to solve environmental problems.

As the scientific and technical arm of the Agency, The Office of Research and Development is responsible for advancing the state of knowledge about the environment such that critical issues and questions can be addressed and answered effectively, based on the application of state-of-the-art science and technology. In the years since 1970, The Office of Research and Development has produced manifold increases in the data base from which environmental decisions are made and in the sophistication of the understanding which has provided the basis for decisions.

This volume represents our effort to take stock of scientific advances in research pertaining to the coastal environment since the inception of the Agency and to gauge what progress has been made and what remains to be accomplished. The essays in this volume present a range of perspectives on the subject, from the vantage points of the scientific and technical disciplines which have been carrying out relevant research. The points of view represented are varied and sometimes conflicting. But scientific progress depends on just such diversity. The authors at times have speculated about emerging problems and research needs. Such attempts require extrapolation based upon informed scientific judgment. The outcome of that process must, in the final analysis, be recognized as opinion and not fact.

PREFACE

This publication is one of several monographs prepared to commemorate the U.S. Environmental Protection Agency's tenth anniversary. It is fortuitous but appropriate for the Agency to begin publication of a monograph on "The Impact of Man on the Coastal Environment" in 1980, which was officially designated by the President as "Year of the Coast." The Environmental Protection Agency was created in 1970 by Executive Order to focus, within one agency, all regulatory-related pollution research conducted by the federal government. Such a mission includes research on the coastal environment, where human land-based activities often interface with productive coastal waters. In the context of this monograph, the coastal environment includes estuarine and marine waters.

The purpose of the monograph is to provide scholarly discussions of some major coastal problems that were addressed during the past decade. The authors of these chapters were chosen because of their expertise in specific aspects of coastal ecology and their accomplishments as marine scientists. They represent various academic institutions and the National Oceanic and Atmospheric Administration (NOAA) that are concerned with research in the coastal environment. Each was charged to describe the issues, their significance and context, to develop the state of knowledge, noting progress through the decade, and to project long-term research needs for the future. The papers were to be more interpretive than review-oriented. Although the problems discussed are of interest to the Environmental Protection Agency, the authors were asked to base the presentation on their perspective, regardless of the Agency's involvement or actions.

The Environmental Protection Agency's coastal research activities are distributed among three environmental research laboratories located at Corvallis, Oregon; Narragansett, Rhode Island; and Gulf Breeze, Florida. Each laboratory conducts research in the coastal zone in which it is situated and participates in national research programs. Scientists in these laboratories develop and interpret a scientifically sound and legally defensible data base in response to the Environmental Protection Agency's regulatory needs. Research activities vary from developing laboratory test protocols to evaluating the effect and fate of toxic substances in coastal waters and mounting full-scale laboratory-field assessment of the impact of pollutants on specific ecosystems. Activities during the past decade have included such environmental problems as the release of Kepone, a highly toxic pesticide, into the James River and adjacent coastal area; ocean disposal projects in the New York Bight, Atlantic Ocean, and Gulf of Mexico; assessment of the oiling of south Texas beaches by crude oil from the Ixtoc oil well blowout in Mexican waters and other oil spills; and assessment of the effects of various industrial and municipal effluents on coastal systems.

The Environmental Protection Agency's coastal research efforts, although a small part of the total federal commitment to marine research and a small part of the overall effort in this country, are uniquely tied to regulatory mandates from Congress. Environmental Protection Agency scientists involved in regulatory research find it necessary and desirable to communicate with the marine science community, which includes other agencies. Recent legislation, such as the National Ocean Pollution Research and Development Planning Act (PL 95-273), encourages communication among administrators as well as scientists. Under PL 95-273, NOAA was given responsibility for preparing the plan in concert with the Environmental Protection Agency and other agencies.

That law required an assessment of national needs and problems pertaining to ocean pollution research and development and the development of a national plan for coastal pollution research. It also solicited policy recommendations concerning ocean pollution research. An interagency committee was formed to develop a comprehensive 5-year ocean pollution research plan. To that end, workshops involving representatives from public and private sectors met and contributed to the plan. Various subcommittees and workshops sponsored by the interagency committee identified the following major research concerns for the marine area:

- Description and measurement of critical functional components of undisturbed and perturbed ecosystems.
- Measurement of rates and interactions associated with processes and fluxes within individual organisms and in major ecosystems.
- Determination and evaluation of importance of effects (both short- and long-term) on components, processes, and fluxes that constitute significant alterations in organisms and ecosystems.
- Definition of the assimilative capacity for degradable materials in relation to other uses of the marine environment.
- Determination of effects of long-term, low-level, chronic pollution resulting from spills, production, and operational discharges in development of predictive models.
- Improvement of our ability to choose waste treatment strategies and outfall sites. Evaluation of nutrient characteristics of particular in-shore marine ecosystems, separating natural from human-made variations, and accurate determination of the degree and persistence of change that any proposed municipal discharge is likely to produce.

The list of future research needs also reflects, and in some instances strengthens, research efforts started in the past decade in response to pollution incidents that occurred in the coastal zone as well as continued efforts to maintain and improve the ecological "health" of the zone. Subjects for the chapters of this monograph were selected to highlight either events that underscore pollution problems or research efforts that emerged from them. Definition of the assimilative capacity of coastal waters to human wastes (the subject of the first paper) has received much attention, particularly during the latter part of the decade. Papers dealing with the Chesapeake Bay and the New York Bight were chosen to illustrate both the problems and progress in pollution control of valuable coastal areas and may serve as examples of sites where capacity to assimilate specific pollutants has been exceeded. Discussions of the impact of specific pollutants, such as nutrients, oil, and toxic substances, also are presented.

It is appropriate, as a starting point, to examine the assimilative capacity of the oceans (Paper 1, *The Oceans and the Wastes of Human Societies*). In a sense, this sets the stage for the following papers. Assimilative capacity is defined as the amount of a given material that a water mass can absorb without resultant unacceptable impacts, be they upon living organisms or nonliving resources. This amount is determined by titration of the polluting substance and becomes evident at an endpoint. (Pollutant concentrations that result in an effect before an endpoint is reached are referred to as checkpoints.) Radioactivity is used as an example in which unacceptable amounts of ruthenium-106 in the seaweed *Porphyra* near the Windscale Processing Plant in the United Kingdom constitute an endpoint. Examples are given of wastes in the U.S. coastal waters in which the assimilative capacity is not fully reached and others in which endpoints indicate that the capacity is nearing saturation. The role of monitoring in assessing assimilative capacity and the use of physiological responses as indices of adverse effects are discussed.

The Chesapeake Bay has undergone a period of exceptional change during the past decade (Paper 2, *Pollution in Chesapeake Bay: A Case History and Assessment*). Some of this change was caused by the introduction of human pollutants—sewage, nutrients, heat, oil and chemical spills, and toxicants. The actual and potential effects of these pollutants on the Bay are discussed. An evaluation of events following the release of Kepone, a persistent chlorinated hydrocarbon, in the James River is presented with documentation of effects on human health and aquatic species. Although most of the Kepone originally released is buried by recent sedimentation, much of the biota remains contaminated.

The Chesapeake Bay Program, a research endeavor sponsored by the Environmental Protection Agency, emphasizes toxic materials, submerged aquatic vegetation, overenrichment, and improved management of water quality. This study is about three-fourths completed and involves about 60 principal investigators at 30 agencies and institutions. The Bay research program, as well as other recent studies, regards the Chesapeake along with its tidal tributaries as a single entity with physical, chemical, and biological continuity—the total Bay is treated as an ecosystem. This treatment was pointed out as an advance for the decade. An assessment of pollution problems of the Bay indicates new and improved state laws, stronger management involvement, and increased focus for more sophisticated research, all of which may have positive effects. However, population growth continues with attendant loading and introduction of exotic toxicants and other materials which continue to plague the Bay.

The New York Bight is an example of a coastal area where human influences are functions of striking increases in population density and energy usages (Paper 3, *Pollution in the New York Bight: A Case History*). In earlier times, solid wastes were dumped on lower value lands to create new land; garbage and refuse were dumped in the inner Bight, and sediment and sewage sludges were dumped nearby. The discharge of waste in the Bight area is still a concern and has received much attention during the past decade. Modifications of the Bight ecosystem are difficult to document because of limited measurements and large variability in responses to natural environmental fluctuations. Several environmental issues with attendant impacts during the past 10 years are discussed and include bathing water quality, oil spills, and dredged material. In addition, environmental crises (real or imaginary), such as beach pollution by sewage sludge and bioaccumulation of dredged material, are presented. Several limitations to statutes and regulations are discussed in relation to pollution control and management of resources in the Bight. In general, much was accomplished during the 1970s toward identifying and understanding the causes and effects of marine pollution in the New York Bight. It is suggested that in the future, effects of pollution in the Bight be evaluated at the ecosystem level and, if ecological effects are deemed unacceptable, remedial action be taken without necessarily attempting to blame or control any single chemical or type of waste.

Domestic sewage contains nutrients that, when discharged into the coastal environment, have potential for detrimental effects (Paper 4, *Man's Impact on the Coastal Environment: Nutrients in the Marine Environment*). Even when sewage receives secondary treatment to reduce biological oxygen demand before being discharged, the treatment does not remove essential plant nutrients, primarily nitrogen and phosphorus. Occurrence of these conservative elements in domestic sewage and the role of sewage in coastal ecology are discussed. The Hudson Estuary is used as an example of how nutrients introduced into a river or estuary might adversely affect "downstream" coastal waters. Excessive turbidity limits stimulation of phytoplankton photosynthesis in the Hudson Estuary, but nutrients from sewage discharges into the estuary at least partially nourish photosynthesis over a wide area in the New York Bight. The Hudson Estuary also was used to illustrate that the distribution of a conservative element can be predicted readily for steady-state conditions, but details of distribution and mechanisms that control and produce observed distributions are more difficult to understand. Difficulties encountered in making steady-state predictions are presented. These include errors in the use of the total volume versus the freshwater volume of an estuary and the assumption for dilution. The necessity of considering the coastal system as a whole in calculating assimilative capacity is highlighted.

Toxic organics, including pesticides, compounds associated with petroleum or its derivatives, and industrial compounds are introduced into coastal waters and can adversely affect the biota and environment in which they live (Paper 5, *Impact of Toxic Organics on the Coastal Environment*). This chapter focuses largely on pesticides and selected organics used in industrial processes. The banning of DDT in 1972 led to the development and use of a variety of new chemical and biological pesticides. In many instances, these compounds were more specific and less persistent than the organochlorines used previously. Scientific expertise for development and application of tests to measure fate and effects of toxic organics progressed during the decade from simple static tests, with mortality as the only effect criterion, to chronic evaluations, with growth, reproduction, behavior, and other sublethal criteria of effect.

As the assessment methods became more complex, so too did the types and chemistry of the agents themselves. There emerged a group of materials and synthetic chemicals known as "third-generation pesticides" that are insect hormones or mimic the actions of hormones that control growth or metamorphosis. New toxicological and exposure assessment techniques were developed to assess the effect of these new compounds on coastal species. Sublethal effects of these compounds on marine species, especially crustaceans, are described. Mention is made of the need to expand from controlled laboratory tests that rely on conventional techniques to detect low concentrations of pollutants to tests that measure biological response in the environment.

Large amounts of petroleum are produced and transported in the coastal zone, and these activities combine to focus a major impact on this environment (Paper 6, *Impact of Oil on the Coastal Environment*). The contribution of petroleum products to coastal areas through production, transportation, river runoff, and municipal and industrial wastes, as well as the chemical composition of various mixtures of oil, is discussed. Evidently, biological and aesthetic damage by chronic or acute release of petroleum into the environment is a function of the weathering of the oil, which includes such factors as evaporation, emulsification, photochemical oxidation, and biodegradation. The biological effects of oil are presented in a matrix, one side of which represents the level of biological organization (bacteria to fish), and the other, individual petroleum compounds, various fuel oils, and crude oils. Data are given on the toxicity of oil compounds to species and communities. Community studies with plankton in open ocean waters showed that population structure and succession patterns of plankton may be more useful as measurements of stress than are metabolic functions. Information on the kinds and concentration of petroleum in the environment has increased significantly during the past decade as a result of increased petroleum activity and rapid development of analytical instrumentation. Future research on this subject should include an understanding of the relationships between the flow of petroleum carbon and the flow of synthetic carbon.

In my opinion, the authors have met the charge previously stated and produced a publication that will be of value to the scientific community. On behalf of the Environmental Protection Agency, I thank them for taking time from busy schedules to participate in our Decade Project.

Thomas W. Duke
Editor

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THE OCEANS AND THE WASTES OF HUMAN SOCIETIES

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INTRODUCTION

The possible loss or restricted use of marine resources as a consequence of pollutant introduction has been recognized for the past three decades. This understanding developed when the use of nuclear energy was in its infancy and the possibility existed that highly toxic, artificially produced radionuclides could enter the atmosphere and the oceans. Scientists from several countries sought to define the acceptable levels of these radionuclides that the oceans might accommodate without jeopardizing public health or the integrity of marine ecosystems. Discharges of such materials have been regulated using the most reliable scientific information with the result that the world ocean does not appear to have dangerous levels of radioactivity.

During these last 30 years, a series of catastrophic events has identified other polluting materials entering the marine environment. Perhaps most notorious is the Minimata Bay incident. Mercury and its compounds from the Chisso Chemical Corporation, which manufactures plastics and industrial chemicals, were discharged in wastes to Minimata Bay and entered the marine food chain. Fishermen, their families, and their pets ingested methyl mercury chloride and were afflicted with a serious neurological disease that caused over a hundred mortalities and a greater number of morbidities. The disease first became apparent in 1953, and the active agent was identified in 1963 as methyl mercury chloride. Curiously, this compound is the dominant natural form of mercury in marine fish. Although the financial losses to the Japanese are difficult to estimate, they appear to be in the hundreds of millions of dollars.

Kepone, a halogenated hydrocarbon used as a pesticide, was promiscuously released by a chemical manufacturing concern to the James River, which drains into southern Chesapeake Bay. This activity was discovered after workers in the production plant became ill through exposure. The carcinogen contaminated the fish and shellfish in the estuarine environment, resulting in a ban against their commercial harvesting. Losses of hundreds of millions of dollars to the fishing industry are estimated.

Thus, as a result of scientific intuition and catastrophes, a large number of pollutants have been identified within the oceans (NOAA, 1979). This awareness, coupled with monitoring activities, has reduced the possibility of other catastrophic events. Few of these identified pollutants cause any apparent damage to the living or nonliving resources of the sea or to humans who consume fish and shellfish or are exposed in recreational areas. Regulatory actions have reduced the release of other pollutants to the oceans.

Our concerns with marine pollution are a part of a much larger and perhaps much more important problem—a quantification of the ability of the oceans to accept a portion of the wastes of human societies. The peoples of the world utilize about three billion tons per year of minerals, food, and forest products. If this tonnage were compressed into a cube, each edge of the cube would be a kilometer long. (In addition, about 20 billion tons of carbon dioxide are released to the atmosphere through the combustion of oil, coal, and other fossil fuels.) Since the solid and liquid materials are generally not accumulated but are disposed, the identification of sites to accommodate these wastes is of paramount importance, especially if we wish to maintain environmental resources. The use of the oceans as disposal space has been accepted throughout history. Yet extreme care must be exercised in regulating the amounts and types of discharges such that *the resources of the oceans are kept in renewable states.*

Previous investigations of marine pollution problems provide guidance about the possible disposal of benign and toxic wastes to the sea. Clearly, economic and social considerations are involved. But the development of scientific protocols for assessment of assimilative capacities of marine waters can proceed upon the basis of past experience. It is with this view in mind that I will review some of the more important events in marine pollution and in marine chemistry, and I will then present some concepts used in determining assimilative capacities.

THE SPRINGBOARD — RADIOACTIVE POLLUTION

In the early 1950s, many scientists were concerned about the ability of the oceans to accept artificially produced radionuclides and especially about the biological effects of radiation. The mood of the period is well expressed in the following paragraph (Revelle and Schaefer, 1957):

> Among the variety of questions generated by the introduction of radioactive materials into the sea, there are few to which we can give precise answers. We can, however, provide conservative answers to many of them, which can serve as a basis of action pending the results of detailed experimental studies. The large areas of uncertainty respecting the physical, chemical, and biological processes in the sea lead to restrictions on what can now be regarded as safe practices. These will probably prove to be too severe when we have obtained greater knowledge. It is urgent that the research required to formulate more precise answers be vigorously pursued.

The regulatory measures that limit the introduction of radionuclides to the oceans have been formulated over the past decades with the goal of protecting human health. Perhaps the largest amounts of radioactivity introduced today come from the nuclear reprocessing plant at Windscale, United Kingdom. The materials enter the Irish Sea through a pipeline from the various facilities. United Kingdom environmental scientists have developed protocols to regulate such discharges on the basis of the "critical pathways" technique. Herein, radionuclides that are ingested through the consumption of algae, fish, and shellfish or the cumulative gamma radiation of which is emitted from sediments in beach areas and which may achieve levels capable of jeopardizing human health are monitored. The surveillances are carried out on the consumed foods or their basic components and at the beach areas of concern. Supported by assessments prepared by the International Commission on Radiological Protection (ICRP), permissible body burdens of given radionuclides have been evolved based upon their toxicity. A "critical population" is identified consisting of those individuals receiving the highest radiation dose rates, either from food consumption or from beach exposure. The acceptable levels are conservative, with many built-in maximizing assumptions such as those of a lifetime consumption of the radionuclide in question and the proviso that the living materials eventually eaten come only from the area adjacent to the Windscale outfall.

Several classical cases of pollutant management have evolved. The first involved the release to the oceans of ruthenium-106, which is accumulated by the algae *Porphyra*. A small population in south Wales uses the seaweed as a food in the preparation of a pudding called laverbread. Although the pathway is now essentially dormant because the consumers no longer receive their algae from the Windscale area, monitoring is nevertheless still performed (Hunt, 1979). Figure 1 illustrates the results of monitoring activities for this radionuclide.

The laverbread story illustrates effectively the "critical pathways" approach formulated in the United Kingdom. The guiding principle involves the identification of the pathways by which a radioactive substance introduced to the environment can return to the population. In general, for the radioactive isotopes of a given element, only one or two pathways are of importance. Of further importance is the identification of the critical group within a given population that receives the highest dosage of the radionuclide. United Kingdom policy explicitly attempts to control the release of radioactive nuclides to the environment on the basis of protecting the most highly exposed individual (Preston and Mitchell, 1973).

The population exposed to ruthenium-106 through laverbread consumption numbered 26,000. The critical group consumed about 160 grams (5.6 ounces) of the product per day; exposure estimates have been based upon studies of this very small subpopulation. The *Porphyra* that accumulated ruthenium-106 from the nuclear activities at Windscale comprised only a part of the seaweed used by the critical population, although there were times when only the Windscale *Porphyra* were utilized. In estimating total exposures of the critical population, the model assumes that no dilution of the Windscale *Porphyra* with seaweeds containing no ruthenium-106 has taken place. The monitoring program covered sites where the seaweed could potentially be harvested at distances up to 40 km (25 mi) from Windscale. At the beginning of the monitoring program, the dilution of Windscale seaweed with that from other areas could be estimated, since all of the *Porphyra* was shipped by rail. The shipping records were available up until the late 1960s, after which the use of rail transport declined. Subsequently, the monitoring of the laverbread itself became necessary.

External exposure of the public to gamma-emitting radionuclides that became associated with particulate phases and subsequently entered the beach areas was also recognized. The nuclides of concern in the past have been zirconium-95 and niobium-95 (Preston and Mitchell, 1973). The critical population for external exposure was revealed to be one salmon fisherman, whose activities were confined to a single beach at one estuary where the silt and the clay minerals containing the radioactive substances accumulated. This individual spent some 300 hours per year in his pursuit of the salmon. This rather unique case emphasizes the need for social as well as scientific studies.

Another example considers public radiation exposure from radiocesium, which is highly concentrated in fish and shellfish. The highest exposure to members of the critical population, based upon their consumption of these marine products, was 31 percent of the ICRP-recommended dose limit in 1977 (Hunt, 1979).

The significant lesson from these British activities is that highly toxic materials can be released to the marine environment and can potentially return to impact human health. With appropriate models and confirmatory monitoring procedures, risk to the most susceptible individuals can be minimized. Further, of great significance is the public dissemination of all information and data regarding the radioactive releases. Table 1 shows the important releases from Windscale. Here, discharge limits for individual or collectives of radionuclides are given and the percentage of the limit utilized in 1977. Such data are issued on an annual basis.

MERCURY AND SOCIETAL REACTION TIMES

In 1963, following the Minimata Bay epidemic and a similar outbreak in Niigata, Japan, where chemical factories discharged spent mercury wastes into natural

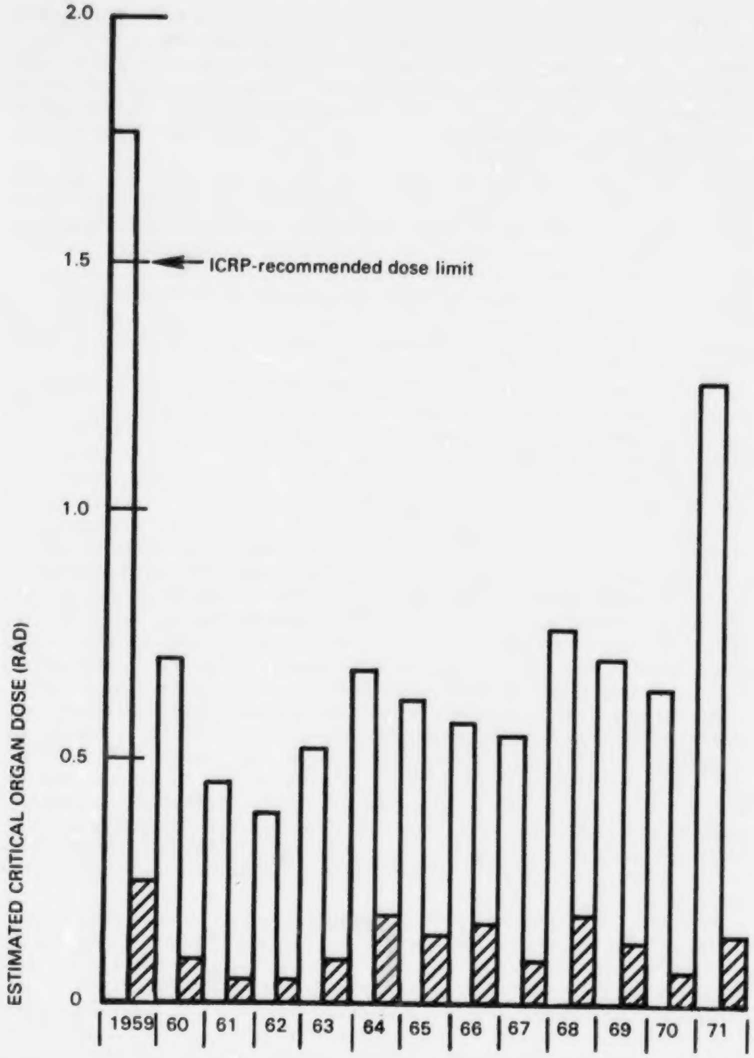

Figure 1. Radiation exposure from consumption of laverbread. Radiation exposure of the GI tract lower large intestine from the consumption of laverbread at 160 g/d. Unshaded blocks-dose-rate based on laverbread manufactured from contaminated weed only; shaded blocks-dose-rate on laverbread manufactured from contaminated weed diluted by the addition of clean weed. (Preston and Mitchell, 1973).

Table 1. Principal Discharge of Liquid Radioactive Waste from Windscale, 1977 (Hunt, 1979)

Radioactivity	Discharge limit (annual equivalent), Ci	Discharge		
		TBq	Ci	000f limit utilized
Total beta	300,000	7,132	192,768	64
Ruthenium-106	60,000	816	22,053	37
Strontium-90	30,000	427	11,534	38
Total alpha	6,000	46	1,241	21

waters, methyl mercury chloride was identified as the active toxin. During the same period in Sweden some disastrous impacts upon fish and wildlife through their consumption of organic mercurial pesticides, often used as seed coatings, alerted some scientists to study the environmental chemistry of mercury. From such investigations, it was established that uncontaminated fish had a mercury burden primarily in the form of methyl mercury.

The first group to evaluate systematically the risks in the consumption of fish containing mercury was appointed in 1968 by the Swedish National Institute of Public Health in conjunction with the Swedish Board of Health and the Swedish National Veterinary Board. The group assessed the toxicological evidence from the Japanese epidemics and the fish-eating habits of both the Japanese and Scandinavian populations. Some Swedish individuals who consumed large quantities of fish and who had no symptoms of Minimata Bay disease had mercury concentrations in their hair and blood similar to those of the Japanese who had shown neurological symptoms of the disease. The results emphasized the varying sensitivities of individuals to methyl mercury poisoning and the different eating patterns of populations from different countries. The average daily consumptions of fish per day in Japan and Sweden are 84 and 56 grams (3 and 2 ounces)/day, respectively, while in the United States the value is 17 grams (0.6 ounces)/day. The evaluation by the Swedish Group (Anonymous, 1973) indicated that 10 percent of the Swedish population might carry the maximum tolerable level in their bodies through the consumption of fish containing 0.5 ppm of mercury (wet weight). This amount has not yet produced Minimata Bay disease. A safety factor of ten is built into the calculations. The limit of 0.5 ppm of mercury in fish, or modifications of it, has been adopted not only by Sweden but by many other northern European countries and by the United States.

The important lesson from the mercury tragedy is that scientists can reach an understanding of a critical pollution problem in the coastal zone and can propose remedial action in decades. It took a bit over two decades for the Japanese government to halt the discharge of mercury into the coastal zone and for other countries to define acceptable mercury levels in seafoods. The levels of mercury in fish from the Baltic Sea were markedly reduced when the Swedish government, acting upon the advice of its scientists, banned the discharge of mercury wastes from chemical plants.

DDT AND THE INTEGRITY OF ECOSYSTEMS

Whereas the protection of human health has been the goal of regulating the inputs of radioactive substances and mercury to the marine environment, another criterion for curbing discharges developed after DDT and other chlorinated hydrocarbon biocides were broadcast about the surface of the earth beginning in 1946. This criterion was the maintenance of the integrity of ecosystems. Several scientists recognized the potential deleterious impacts of DDT on nontarget organisms very early after its initial use. In 1946, Clarence Cottam and Elmer Higgins of the U.S. Fish and Wildlife Service wrote:

> From the beginning of its wartime use as an insecticide the potency of DDT has been the cause of both enthusiasm and grave concern. Some have come to consider it a cure-all for insect pests; others are alarmed because of its potential harm. . . .DDT, like every other effective insecticide or rodenticide, is really a two-edged sword; the more potent the poison, the more damage it is capable of doing. . .the most pressing requirement is a study to determine the effects of DDT as applied to agricultural crops on the wildlife and game dependent upon an agricultural environment. About 80 percent of our game birds, as well as a very high percentage of our nongame and insectivorous birds and mammals are largely dependent upon an agricultural environment. In such places application of DDT will probably be heavy and widespread; therefore, it is not improbable that the greatest damage to wildlife will occur there. Because of the sensitivity of fishes and crabs to DDT, avoid as far as possible direct application to streams, lakes and coastal bays.

Subsequent events have confirmed the observations and predictions of Cottam and Higgins. For instance, one of the impacts upon marine ecosystems was the reproductive failure in the brown pelican population on Anacapa Island, off the California coast, from 1969 to 1972. The accumulation of DDT and its degradation products, primarily DDE, by marine organisms that were the pelicans' food initiated the problem. The source of the DDT was allegedly the wastes from a chemical manufacturing plant in Los Angeles. The result was the production of thin egg shells that broke easily (Risebrough, 1972).

The general use of DDT and other chlorinated hydrocarbon pesticides was restricted in the early 1970s by the United States and many northern hemispheric countries. Of importance in the DDT story is a criterion for the outlawing of its use, its impact upon nontarget organisms, and the loss of integrity of ecosystems, both on land and in the sea.

THE MONITORING MODE

Over the past three decades a large number of polluting substances have been identified as entering the marine environment (NOAA, 1979). In general they can be classified into nine groups:

1. The synthetic organic chemicals, including such halogenated hydrocarbons as DDT and its degradation products, Kepone, the polychlorinated biphenyls, and low molecular weight halocarbons such as carbon tetrachloride and chloroform.
2. The oxidation products from the chlorination and ozonation of waste and cooling waters. These substances result from the interactions of bromine and chlorine with organic molecules. Chloroform and chlorophenols are among the products so far identified.
3. Artificial radionuclides from the nuclear fuel cycle and from nuclear weapons testing. This set includes the fission products (strontium-90, cesium-137), the fuel materials and their alteration products (plutonium-239, plutonium-238, americium-241), and the induced activities (manganese-54, cobalt-60).
4. Biostimulants, the plant nutrients such as compounds of nitrogen, phosphorus and silicon, trace metals, and dissolved organics that cause increased plant productivity. As a consequence, alteration of the plant community structure and possibly eutrophication of the waters can come about.
5. Microorganisms, the agents of human and faunal disease, bacteria and viruses.
6. Trace metals that can inhibit plant and animal productivity or can make seafoods toxic. Attention has been drawn to lead, cadmium, copper, mercury, and arsenic, among others.

7. Fossil fuel compounds, the hydrocarbons and other organic compounds of petroleum, natural gas, coal, and oil shale. In addition there are the products of their combustion and transformation through chemical, photochemical, microbial, or metabolic actions.
8. Litter, a collective of materials that can be defined as any anthropogenic or natural solid product that is out of place in the marine environment. The product may be composed of plastic or other synthetic organic materials, glass, wood, petroleum components in the form of tar or grease balls, and natural articles resulting from improper disposal.
9. Dredged materials that may contain some of the pollutants listed above and large volume wastes such as sewage sludges and industrial discards.

Clearly, any given coastal water body will not be subject to insult by all collectives of these polluants. Surveillance activities usually involve only a very few substances. The tactics of monitoring depend not only upon the pollutants of concern but also upon economics. Analyses of seawater are expensive. Assaying accurately for most of the pollutants, especially those in extremely low concentrations, taxes the resources of even the best analytical facilities. The collection of large volumes of water without contamination and the subsequent assay schemes require sophisticated sampling equipment and analytical instrumentation. An additional difficulty with water assays is that they give instantaneous levels, whereas often an average or integrated value over a longer time period is desired.

The major surveillance programs have utilized sentinel organisms or sediments, usually directed at members of only one or two of the pollutant sets. The organisms integrate the pollutant exposure levels for periods of days to years while the sediments can often reveal exposure levels averaged over a year. In both cases there are restricted groups of pollutants amenable to measurement. For example, Holden (1973) directed an international cooperative study of organochlorine and mercury residues in wildlife, utilizing mussels, herring, pike, and eel as the sentinel organisms and the eggs of heron, eider, tern, and pelican. Twenty-six laboratories from 12 countries were involved. Butler et al. (Butler and Schutzmann, 1978 and Butler et al., 1978) have utilized estuarine mollusks and fish to monitor 20 organochlorine and organophosphate pesticides and polychlorinated biphenyls in U.S. programs beginning in 1965.

Perhaps the most extensive program has involved the use of bivalves to survey annually the levels of chlorinated hydrocarbons, artificial radionuclides, metals, and fossil fuel compounds at somewhat over a hundred stations around the coast of the United States (Goldberg et al., 1978).

The collections included the mussels *Mytilus edulis* and *M. californianus* and the oysters *Crassostrea virginica* and *Ostrea equestris*. Populations of sufficient size were sampled such that only an insignificant number of the members were used. Organisms of uniform size were taken, usually 5 to 8 cm (2 to 3 in) long, although larger oysters were often utilized. Where possible samples were collected from rock, sand, or mud environments. Pilings or metal buoys were avoided to minimize uptake by bivalves of paints, creosote, or other materials that might have been applied to the substrates. Immediately after collection, the samples were placed in plastic bags (for heavy metals or radionuclide assay) or aluminum foil (for petroleum hydrocarbons and synthetic organic assay) and frozen. Samples were airshipped to the participating laboratories in styrofoam-lined cardboard shipping containers with dry ice.

A single scientist operating from a camper made the collections from June to December starting in southern California and traversing the west coast of the United States. Then a trip across the country took him to the east and gulf coast sampling sites. The use of a single scientist provided uniformity in sampling and preservation techniques.

Collection costs in 1978 averaged about $1000 per station, including air shipments to laboratories and the maintenance of library specimens from all stations.

Analytical costs were: hydrocarbons, $1000 per sample; radionuclides, $550 per sample; and heavy metals, $50 per sample. The pollutants analyzed are listed in Table 2.

The program has identified varying degrees of pollution in U.S. coastal waters. PCB levels at New Bedford Bay, Massachusetts, were so high that bans were placed upon commercial fishing activities. Here the mussels had the highest PCB contents measured in the program. High levels of DDT and its degradation products were found in mussels taken from waters between San Francisco, California, and San Diego, California. The source of these pesticides was a manufacturing plant in Los Angeles, California, between the two cities cited above. Wastes from the plant were discharged through a sewer outfall to the oceans. There are a number of heavy-metal "hot spots." Elevated levels appear in mussels from the New York-New Haven area. The highest concentrations of copper and cadmium occur in organisms collected in New Haven Harbor, Connecticut.

Table 2. Pollutants Analyzed in U.S. Mussel Watch

Heavy metals:	Lead, cadmium, silver, zinc, copper, and nickel. Mercury was added in 1978.
Radionuclides:	Pu-239+240, Pu-238, Am-241, and Cs-137.
Halogenated hydrocarbons:	p, p'-DDE and p, p'-DDD PCBs (1254 and 1260).
Petroleum hydrocarbons:	Naphthalene, methylnaphthalenes, C-2 naphthalenes, C-3 naphthalenes, phenanthrene, methylphenanthrenes, C-2 phenanthrenes, fluoroanthene, pyrene, chrysene/benzanthracene/triphenylene, benzopyrene/perylene/benzofluoroanthene, dibenzothiophene, and methyldibenzothiophene.

Coastal marine sediments, especially anoxic ones, contain historical records of pollution. Usually, chronologies are developed on the basis of radiometric measurements, such as lead-210 or cesium-137 assays, or by the counting of varves. Heavy metals, artificial radioactivities, and halogenated hydrocarbons have been studied usually over periods of about a hundred years or less (Goldberg et al., 1978). These historical records have been of greater use in studying the effects of regulatory or remedial measures to reduce pollutant fluxes to the marine system than as monitoring tools.

An example of sedimentary records of pollutants and associations with source functions evolves from some recent studies at Lake Michigan. The area surrounding the waters is a site of intensive agricultural and industrial activity. Natural and human-induced combustion processes are responsible for the inputs of a variety of materials to the lake. Some are toxic to living organisms; others are benign. The flow of such materials can be followed by the charcoals produced by the incomplete combustions of woods, coals, and petroleums. The morphologies and surface characteristics of these charcoals (soot) are indicative of the different burning processes.

Increased fluxes of the charcoals are evident in the sedimentary strata deposited after 1900. Charcoal concentrations rose constantly until about 1968, at which time they reached a maximum (Figure 2). The tin, chromium, nickel, lead, copper, cobalt, cadmium, zinc, and iron concentrations in the sediments displayed similar profiles as a function of age of deposition. The maximum appears to be related to the period when improved control devices that remove fly ash from the stack gases issuing from energy-producing and industrial facilities were installed. After 1968 lower levels of particulates are reported in the atmosphere adjacent to Lake Michigan. In the deposit, fly ash particles increased in numbers in a fashion similar to that for charcoal and heavy metals.

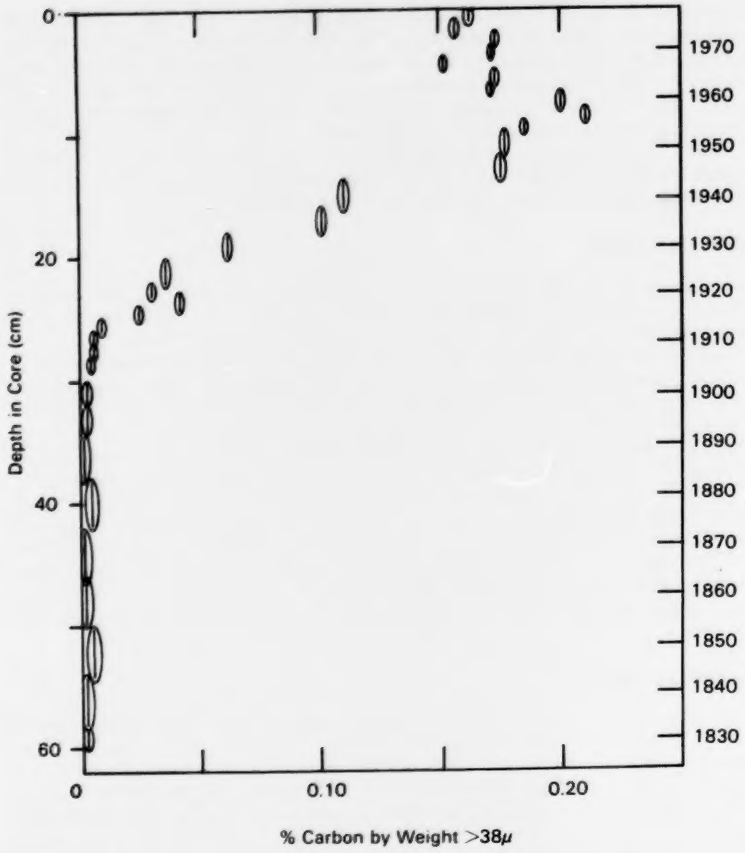

Figure 2. The depth profile of elemental carbon (charcoal) in a Lake Michigan sediment. The ages were determined by Pb-210 geochronology.

On the other hand, one component of the sediments, quartz, the source of which is dominantly natural, decreased markedly in concentration in the recently deposited strata, as compared with pre-industrial revolution values. The fall-off in concentration amounted to about 20 percent and resulted from the increased fluxes of fly ash and associated debris to the deposit site from industrial activities.

There are a continually increasing number of studies illustrating similar environmental histories of pollutants based upon records in sediments. Of importance is the knowledge that the historical records are available, especially for those substances that are recognized as toxic well after the time of their initial introduction to the environment. In addition, knowledge about the persistence of organic materials in the marine environment can be drawn from their persistence in the sediments.

THE STEPPING STONES

In the past three decades, during which the problems of marine pollution have been identified, knowledge about the oceans has grown dramatically. A part of this knowledge has emerged from pollution studies. For example, the importance of atmospheric transport of organic materials from the continents to the oceans was acknowledged following studies on the dispersal of DDT and the polychlorinated biphenyls. The prevalence of methylated species of metals and metalloids in seawaters, sediments, and organisms was recognized from investigations that evolved from the methyl mercury poisoning epidemics in Japan.

Some of the concepts that emerged during this period have guided many pollution studies and have provided stepping stones for developing strategies for the disposal of wastes in the sea. Several of these are considered here.

Bioaccumulation

Some species of organisms have the unique ability to extract from their environment and concentrate polluting materials that can affect their own health or the health of the organisms that consume them, including human beings.

The concentration factors for marine organisms, the ratio of the concentration of a given species in the organisms on a wet weight basis to the concentration in seawater, can rise to levels of hundreds of thousands. For example, ruthenium, one of whose radioisotopes is involved in the laverbread story, has a concentration factor for phytoplankton on the order of 200,000 (Lowman et al., 1971). As a consequence of such enrichments, biological transport is an important factor in governing the distribution of some elements in seawater. Lowman et al. indicate that the diurnal vertical migration of organisms, fecal pellet production, moulting, and death have an overall effect of moving biomass from surface to deeper waters. For pollutants that are bioaccumulated, the dispersion by vertical transport, especially in highly productive waters, can result in their rapid dilution. On the other hand, such mobilization can carry pollutants to the benthos where they can impact upon the communities therein.

Residence Times

It is often essential to know the average period of time that a substance spends in one of the reservoirs of the marine environment—the water, organisms, and sediments, for example—in order to predict the fate of pollutants. The concept of residence time gained momentum during the initial considerations of the disposition of artificially produced radionuclides in the oceans. The marine environment was envisaged as consisting of a number of reservoirs or boxes, various water masses, the phytoplankton, and so on. The transfer of materials from one reservoir to another was assumed to occur through first order kinetics (Craig, 1957).

Of special interest is the concept of biological residence time or half-life. Experiments to determine half-life can develop relationships between an organism and the environmental concentration. There is a wide spectrum of biological half-lives. Smaller marine organisms, including phytoplankton and zooplankton, have biological half-lives on the order of hours (Lowman et al., 1971). For methyl mercury in a human being, the value is about 90 days. Values of years or decades might be expected for transuranics in human bone. Knowing a biological half-life for a substance and estimating future exposure levels can make possible predictions for future body burdens.

Studies of the association of a pollutant with a reservoir, be it an atmospheric wind system or a water body, have emphasized the widespread dispersals of a pollutant in the marine environment and have provided techniques to calculate residence times. A pollutant may be injected into the environment in one country and impact upon another, perhaps even in another continent. Part of the DDT sprayed upon agricultural crops in Africa is transported by the northeast trade winds to the Caribbean. Radioactive debris from the explosion of a Chinese nuclear device in May 1965 was detected at sampling sites in Tokyo and Fayetville, Arizona, during two circumnavigations of the earth. The average velocity of the wind transport was about 16 m/sec (52 ft/sec) in the tropospheric jet streams. The residence time for the debris in the atmosphere was about 2 weeks. Of importance is the delivery of these materials to the ocean system along the paths of the prevailing jet streams.

Speciation and State

The speciation and the state (solid, liquid, gaseous, or colloidal) of an element are important characteristics in governing the residence time and its bioaccumulation. For example, uranium exists in both dissolved and particulate phases in ocean water. The two forms in coastal waters can be identified by their U-234/U-238 ratio (Hodge et al., 1979). Some organisms such as scallops preferably accumulate the particulate form, which is a thousand-fold lower in concentration than the dissolved form. Others such as mussels prefer the dissolved form for uptake. Thus, organismic concentration factors must be related to the particular form in which the element occurs.

Where the various states of an element can be determined with relative ease, the speciation can be a very vexing task. The experimental determinations of the speciation of most elements are quite limited. The species of lead introduced to the marine environment following the combustion of lead alkyls in gasolines may be quite different from the naturally occurring species. As a consequence, their behavior in coastal waters, especially in regard to uptake by organisms, may not be predictable from knowledge about those of lead introduced in the major sedimentary cycle.

Analytical Techniques

In the last half of the 1970s, there has been a minor revolution in our ability to assay trace metals in coastal waters (Bruland, 1980). Improvements both in the sampling of seawaters without the introduction of contamination from the ships, hydrographic wires or sampling devices, and in laboratory analytical techniques resulted in new wisdom about the concentration and distribution of metals in the water column. Concentrations of cadmium, zinc, nickel, and copper were found to be one to three orders of magnitude lower than the values previously reported. Further, it was found that the seawater concentrations of a number of these elements were governed by biological processes. Their abundance profiles, in some cases as a function of depth, appeared to follow those of nutrients such as phosphorus, silicon, and nitrogen.

The improvement in analytical techniques recalls the work of Haber (1928) who in the 1920s reduced the generally accepted values of gold in seawater by about three orders of magnitude. His laboratory precautions and his sampling techniques provided a model for future workers. However, the need for care was somehow overlooked in the following decades until the studies of Claire Patterson at the California Institute of Technology, who, with his co-workers, established new standards in analytical techniques and sampling devices that permitted the accurate determination of lead levels in seawater.

These recent investigations of trace metal abundances have emphasized the importance of societal contributions to the oceans. The coastal waters of industrial areas have received greater fluxes of metals as a consequence of industrial and domestic discharges. But of greater concern is the possibility that marine organisms are sensitive to slight increases of these very low (nanomolar) levels of metals. Anderson and Morel (1978), for example, report that the dinoflagellate *Gonyaulax tamarensis* becomes nonmotile at 0.1 nanomolar additions of cupric iron. Nonmotile cells do not divide or grow larger. These are concentrations that are sometimes found in coastal waters.

Marine Biochemistry

The increased understanding of the uptake, metabolism, and effects of pollutants in the marine biosphere has given us a basis for assessing potential effects and formulating control measures. New definitions of toxicity have evolved and field measurements for toxic effects on some organisms are now possible.

Extensive work has been performed on the pathways of metals through marine organisms, and it has been well summarized recently by George (1980). For phytoplankton and shellfish, the initial uptake of dissolved phases is consistent with passive diffusion, i.e., an initial adsorption to an exposed mucous sheet or cell membrane, followed by diffusion and binding to intracellular components. Metals can also be taken up in particulate form from food, which may be of equal or greater importance than the accumulation of dissolved forms. Some metals are bound nonspecifically to cystosolic proteins. Excessive amounts of a metal may be detoxified in a variety of ways. In some cases they are stored in sulfate or phosphate granules, or directed to the shell, byssal threads, or carapaces.

The studies on the marine biochemistry of halogenated and petroleum hydrocarbons have paralleled those of heavy metals. Copepods enzymatically metabolize petroleum hydrocarbons to hydroxylated forms that are later excreted (Lee, 1975). These organisms can take up dissolved or particulate forms of the hydrocarbons from water or from food. The pollutants concentrate in the livers or gall bladders of fish and are subsequently discharged in the urine or feces (Lee et al., 1972).

THE TITRATION

Over the past three decades, environmental scientists have identified a large number of polluting substances entering the oceans and potentially capable of interfering with public health, the vitality of marine organisms, and the nonliving resources of the sea. Those responsible for the management of the coastal environment have been able to react in decades or less to available information from the scientists. Thus, the experiences with marine pollution and with marine chemistry can act as an information base for considering the abilities of the marine environment to accept some of the large-scale wastes of human beings. Much of the material in the following presentation derives from the deliberations of 70 scientists at the Crystal Mountain Workshop, held in August 1979 (NOAA, 1980).

The assimilative capacity of a marine water body may be defined as that amount of a given material that can be contained within a body of seawater without producing an unacceptable impact, be it upon living organisms or upon the nonliving resources. This amount, essentially determined by a titration of the polluting substances in the discharged material with the water body becomes evident at an endpoint. Pollutant concentrations that are determined before the endpoint is reached are checkpoints. The most extensive set of endpoints for individual pollutants has evolved from artificial radioactivity studies. For example, the unacceptable concentration of ruthenium-106 in the seaweed *Porphyra*, taken from the seawaters adjacent to the Windscale Reprocessing Plant in the United Kingdom, would constitute an endpoint. The 0.5 ppm level of mercury in fish provides an endpoint that protects the heavy fish-eating populations of the world.

With the titration concept, models have been constructed to seek out the assimilative capacities of seawater utilizing existing data (NOAA, 1980). Although such models will be refined with additional data, their construction emphasizes that there is a scientific basis for regulating the discharge of wastes to coastal waters.

The overall conclusion from the Crystal Mountain Workshop is that the waste capacity of U.S. coastal waters is not now fully used. For example, the largest U.S. industrial dumpsite (Site 106 off the coast of New Jersey), which receives about 800,000 m^3 (1,048,000 yd^3)/year of titanium dioxide production wastes, organic chemical wastes, and water treatment materials, is not used to its total assimilative capacity. Here the endpoint is defined as an unacceptable disturbance to the community of marine organisms. The waters of the Southern California Bight have successfully accommodated the waste discharges from the highly industrialized and populated (11 million people) adjacent land areas for the past 20 years without unacceptable effects as determined by studies on the marine plant and animal communities. The most important sources of pollutants are five large municipal wastewater outfalls that discharge about 8 billion l/day (2 billion gal/day). Increased amounts of metals and nutrients are evident in the Bight waters, as well as in the sedimentary records. Amounts of organic particulates have also risen as a consequence of sewage discharges. These organic phases appear to be incorporated into the planktonic food web without impact.

In two of the studied areas, Puget Sound and the New York Bight, there is evidence that the assimilative capacities for some substances may have been reached or exceeded.

Twenty million persons live in the lands adjacent to the New York Bight. It is one of the most intensively used coastal regions of the world. Its assimilative capacity for dumped excavation dirt and construction debris was exceeded in the late 1800s and shoaling of the channels interfered with the passage of ships. Four pollutants were examined for their potential effects upon the Bight: microorganisms, nitrogen-containing biostimulants, polychlorinated biphenyls, and cadmium. Of these, only cadmium appeared to have reached unacceptable levels. For this metal, an endpoint of 5.00 ppb in marine waters has been proposed on the basis that, at this concentration, some oysters accumulate enough of the metal to nauseate human oyster eaters. The highest estimates of cadmium now present in the waters are substantially lower than this amount. Nevertheless, a model using reasonable partition coefficients between shellfish and suspended sediments indicates that organisms growing in heavily contaminated dredge spoils might exceed safe limits for their body burdens of cadmium.

Puget Sound receives about 25 percent of the wastewater from the municipal treatment plant (METRO), which discharges between 470 and 1,279 m^3 (611 and 1,675 yd^3)/day. There is some evidence that the recent toxic dinoflagellate blooms in the central basin may be related to these discharges as may one incident of oyster larvae mortality. In previous years, pulp mill discharges caused depressions in oxygen concentrations, changes in the structure of benthic communities, and toxic effects upon marine organisms. Increased and effective treatment facilities, combined with a leveling off in activity, has resulted in reduced environmental stresses.

ENDPOINTS

In pollution monitoring, the endpoints have usually involved a single substance or a collective of similar substances such as the polychlorinated biphenyls or DDT and its degradation products. Exposure levels in seawaters are generally determined indirectly, primarily through the use of sentinel organisms, although concentrations in waters have been directly determined. For a determination of the assimilative capacity, often the material to be discharged contains a variety of pollutants, the analyses of which would be extremely costly. It appears that impacts of such materials upon the well-being of marine organisms can provide endpoints, especially those upon animal health. Some biological effects are already well established, while others will require further assessment.

Bivalves, especially mussels, have several attributes for use as indicators of general biological stress. First of all, they are cosmopolitan, and often a given species can be found over a wide latitudinal spectrum. Secondly, they are sturdy creatures and can endure considerable physiological or biological stress without mortality. Still, when adaptive mechanisms fail to respond, a measurable impact may be found. A sampling of such impacts taken from Bayne et al. (1980) will be cited in the following paragraphs.

Bayne and his colleagues argue that physiological responses that may be integrated into the ability of the organism to grow have been shown to be effective indices in both field and laboratory studies. This study of "scope for growth" seeks measurements of the changes in feeding rate, absorption efficiency, excretion, and respiration rates and introduces them into a balanced energy equation. A decline in growth potential is a clear sign of a stress. Any statistically valid evidence of a decrease might constitute an endpoint.

Metals in excessive amounts are known to interfere with enzyme systems. For example, cadmium and mercury can displace copper or zinc from metalloenzymes such that the enzyme is rendered inactive. However, in many organisms detoxification activity is a response to metal stresses. This mechanism involves the binding of the metal to the low molecular weight protein, metallothionen. The amounts of the metals bound to metallothionen will increase with increasing exposure levels to a saturation state. At this point, the metals will spill over into the higher molecular proteins in the enzyme pool. When the binding capacity of metallothionen is surpassed, measurable toxic effects occur such as a decreased growth rate. The spillover point then corresponds to the endpoint. Mussels are known to detoxify heavy metals with metallothionen.

The partitioning of metals into lysosomes in kidney, digestive gland, gut, gill, and blood cells is presented as a detoxifying mechanism for mussels (Lowe and Moore, 1979). As the levels of metals in the lysosomes increase in mussels, the lysosomal latency decreases. As a result, hydrolytic enzymes are released into the cytoplasm. Further, as the storage capacity of the lysosome for metals is exceeded, it appears that the metals are also released into the cytoplasm. Thus, studies on the composition of the cytoplasm can be revealing of stress resulting from collectives of polluting metals.

Some empirical measurements seem to reflect the stress syndrome. For example, Jeffries (1972) indicates that clams reflect pollutant exposures through their taurine to glycine ratios. Acute stresses are indicated when the molar ratio is greater than 5. Normal values are 3 or less, while chronic stresses may be described when the ratio varies between 3 and 5. The theoretical basis for this phenomenon has yet to be discovered.

Histopathological techniques show great promise in establishing the relative health of animals (Bayne et al., 1980). Recent investigations with mussels from both polluted and nonpolluted environments have revealed a variety of conditions that in general can be identified by different investigators studying the same samples: (1) hyaline degeneration of the connective tissue of the gills; (2) parasite burdens; (3) increases in the number of mucous secretory cells; (4) gonadal neoplasms; (5) hemopoietic neoplasms; (6) granulocytomas; (7) hemocytic infiltration of tissues; and (8) loss of synchrony in digestive tissues.

Other avenues might be taken to reveal general stress conditions that so far have been only modestly investigated. Such genetic indices as the breakage of chromosomes or chromatids might be useful. So far studies with marine organisms have been carried out with pollutant levels that are generally higher than those ever observed in the environment. Still, such effects might be useful in assessing impacts upon organisms exposed to increased levels of pollution.

THE FUTURE NEEDS

Past activities in marine pollution provide a substantial basis for the consideration of waste disposal in the oceans. Clearly, for any given material awaiting disposal, three options are possible: disposal to the atmosphere, to the oceans, or to land. Each has its advantages and disadvantages based on scientific, social, and economic considerations. For example, the burning at sea of toxic halogenated hydrocarbons and the discharge of the resultant water, carbon dioxide, and hydrochloric acid to the atmosphere appears to be a rational option. The storage of high-level radioactive wastes, where possible retrieval is essential, dictates land sites for appropriate receptacles. Clearly, the marine environment has accommodated domestic and industrial wastes in the past. With increasing affluence in many countries and with an increasing world population, the disposal needs for societal wastes will also increase. What additional information do we need to consider oceanic discharge with the caveat that we maintain the oceanic resources in renewable states?

The simplest answer presupposes increases in our knowledge of the marine environment, chemical, physical, biological and geological, for all contribute to our ability to evaluate the ocean's capacity for accepting wastes.

But some specific problems, often pollutant-specific or site-specific, need to be resolved. Since the impact upon the health of the communities of organisms will determine most endpoints, clearly we require simple, direct indicators of stress. I have pointed out that there are general stress indices already available; still, there remains the identification of specific indices responsive to individual pollutants or classes of pollutants, say, the low molecular halocarbons or chlorophenols.

Some background problems continue to haunt us. What are the long-term effects of low levels of pollutants on life in the sea? What are the synergistic and antagonistic effects of collectives of pollutants or of individual pollutants? What are the amounts and compositions of discharged wastes going to the oceans today? What are the anticipated amounts for the near future?

But also there are scientific developments that will aid in our study of the titrations. Remote sensing may be of great value in studying the dispersion of wastes after introduction and the description of effects, for instance, eutrophication, where one species of algae is replaced by another. Our abilities to analyze smaller and smaller amounts of pollutants and to collect environmental samples without contamination have continually developed over the past years. There appears to be no let-up in this area. Finally, the growing community of political scientists, economists, and sociologists demands an evaluation of external costs in considerations of oceanic discharge, information essential to ascertain whether the oceans can compete with the atmosphere or land in receiving specific societal discards.

REFERENCES

Anderson, D. M., and F. M. M. Morel. 1978. Copper sensitivity of *Gonyaulax tamarensis*. Limnol. Ocean. 23:283–296.

Anon. 1973. Methyl mercury in fish. A toxicologic epidemiologic evaluation of risks. Report from an expert group. Noridsk Hydienish Tidskrift, Supplementum 4.

Bayne, B., et al. 1980. Mussel health. The international mussel watch, December 1978. U.S. National Academy of Sciences.

Bruland, K. W. 1980. Oceanographic distributions of cadmium, zinc, nickel and copper in the North Pacific. Earth Planet. Sci. Letters 47:176–198.

Butler, P. A., and R. L. Schutzmann. 1978. Residues of pesticides and PCBs in estuarine fish. 1972–1976. Pesticides Monitoring Journal 12:51–59.

Butler, P. A., C. D. Kennedy, and R. L. Schutzmann. 1978. Pesticide residues in estuarine mollusks, 1977 versus 1972. National Pesticide Monitoring Program. Pesticides Monitoring Journal 12:99–101.

Cottam, C., and E. Higgins. 1946. DDT: its effect on fish and wildlife. U.S. Department of Interior, Fish and Wildlife Service Circular 11.

Craig, H. 1957. Isotopic tracer techniques for measurement of physical processes in the sea and atmosphere. In: The effects of atomic radiation on oceanography and fisheries. U.S. National Academy of Sciences-National Research Council Publication 551:103–120.

George, S.G. 1980. Correlation and metal accumulation in mussels with the mechanisms of uptake, metabolism and detoxification: biochemical and ultrastructural studies. Abstracts, VI International Symposium on Chemistry of the Mediterrranean, Rovinj, Yugoslavia.

Goldberg, E. D. 1979. Pollution history of some U.S. estuaries. Proceedings of Aquatic Environment in Pacific Region, August 21–23, 1978. SCOPE/Academia Sinica, Taipei, Republic of China. pp. 8–18.

Goldberg, E. D., et al. 1978. The mussel watch. Environmental Conservation 5:105–125.

Goldberg, E. D., J. J. Griffin, M. Koide, and V. Hodge. 1980. The impact of fossil fuel combustion on the sediments of Lake Michigan. Manuscript.

Haber, F. 1928. Das gold im meere. Z. ges. Erdkunde 3:1–12.

Hodge, V. F., M. Koide, and E. D. Goldberg. 1979. Particulate uranium, plutonium and polonium in the biogeochemistries of the coastal zone. Nature 277:206–209.

Holden, A. V. 1973. International cooperative study of organochlorine and mercury residues in wildlife, 1969–1971. Pesticides Monitoring Journal 7:37–52.

Hunt, G. J. 1979. Radioactivity in surface and coastal waters of the British Isles, 1977. Aquatic Environment Monitoring Report Number 3, Ministry of Agriculture, Fisheries and Food, Directorate of Fisheries Research ISSN 0142–2499, Lowestoft, United Kingdom.

Jefferies, H. P. 1972. A stress syndrome in the hard clam, *Mercenaria mercenaria*. J. Invert. Pathol. 20:242–251.

Lee, R. F. 1975. Fate of petroleum hydrocarbons in marine zooplankton. In: Proceedings of the 1975 Conference on Prevention and Control of Oil Pollution, American Petroleum Inst., Washington D.C. pp. 549–553.

Lee, R. F., R. Sauerheber, and G. H. Dobbs. 1972. Uptake, metabolism and discharge of polycyclic aromatic hydrocarbons by marine fish. Mar. Biol. 17:201–208.

Lowe, D. M., and M. N. Moore. 1959. The cytolochemical distributions of their zinc (Zn II) and iron (Fe III) in the common mussel, *Mytilus edulis*, and their relationship with lysosomes. J. Mar. Biol. Assoc., United Kingdom 59:851-858.

Lowman, F. G., T. R. Rice, and F. A. Richards. 1971. Accumulation and redistribution of radionuclides by marine organisms. In: Radioactivity in the marine environment. U.S. National Academy of Sciences. pp. 161–199.

NOAA. 1979. Proceedings of a workshop on scientific problems relating to ocean pollution. Estes Park, CO, July 10–14.

NOAA. 1979. The assimilative capacity of seawater for society wastes. Proceedings of a workshop held at Crystal Mountain, WA, July 30–August 4.

Preston, A., and N. T. Mitchell. 1973. Evaluation of public radiation exposure from the controlled marine disposal of radioactive waste (with special reference to the United Kingdom). In: Radioactive contamination of the marine environment. International Atomic Energy Agency, Vienna. pp. 575–593.

Revelle, R., and M. B. Schaefer. 1957. General considerations concerning the ocean as a receptacle for artificially radioactive materials. In: The effects of atomic radiation on oceanography and fisheries. U.S. National Academy of Sciences-National Research Council Publication 551:1–25.

Risebrough, R. 1972. Cited in: Birds and pollution, an editorial article. Nature 240:148.

POLLUTION IN THE CHESAPEAKE BAY:
A CASE HISTORY AND ASSESSMENT

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POLLUTION IN THE CHESAPEAKE BAY: A CASE HISTORY AND ASSESSMENT

THE BAY

The tidal portion of the Chesapeake Bay system is the largest and most complex estuary in the United States and the most valuable to human interests. It is a drowned river valley system with many tributaries. Density gradients drive the two-layered circulation typical of such estuaries. Detailed description is not appropriate in this summary, since it has been provided in other publications (Chesapeake Research Consortium, 1977; Corps of Engineers, 1974; Corps of Engineers, 1977; Cronin, L., 1967; Cronin, L., 1976; Cronin, L., 1978; Cronin, W., 1971; Kuo et al., 1975; Lippson, 1973; Schubel, 1972).

Table 1 summarizes physical characteristics, the human population of the basin, some of the principal uses, and projections for the future. The tidal Bay system lies in Maryland and Virginia, but the drainage basin also includes portions of New York, Pennsylvania, Delaware, the District of Columbia, and West Virginia. The Bay is complex, highly productive, valuable for many purposes, and subject to rapid expansion in usages, many of which currently cause pollution or create conflicts. The Bay is outlined in Figure 1, and the names of places cited in this review are noted.

The Chesapeake is the site and center of rapid change and vigorous human activity. Most pollution problems are related to the sites of human population and industrial activity; Figure 2 illustrates the locations of people and industry. While this report emphasizes history, it is appropriate to include the best available projection of future distributions in Figure 3. The total population is expected to double, with related increases in all aspects of pollution.

Even for the last decade, it is not feasible to report the full history of pollution pressures, water quality, new learning, legislation, and action by government and the public in a relatively short summary. The more limited purposes of this report are to identify the principal pollutants, note exceptional progress and problems related to

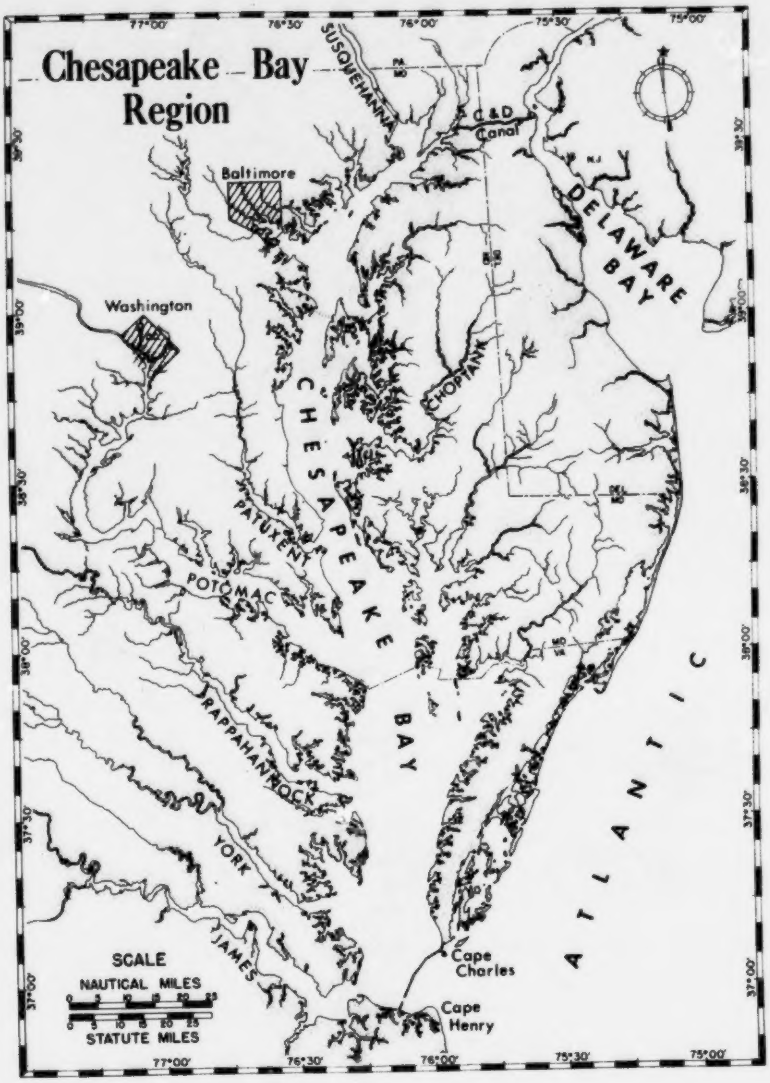
Table 1. Chesapeake Bay—Physical and Biological Characteristics, Selected Usages and Basin Population (Corps of Engineers, 1974; Corps of Engineers, 1977; Cronin, W., 1971; Lippson, 1973)

Physical		
Length:	290 km (180 mi)	
Width:	4-48 km (5-30 mi)	
Depth:	53 m maximum (175 ft) 8.4 m average open Bay (27.6 ft) 6.5 m average including tributaries (21.2 ft)	
Surface area:	6,500 km² open Bay (2,500 mi²) 11,500 km² with tributaries (4400 mi²)	
Shoreline:	13,000 km (8,100 mi)	
Volume:	52 billion m³ for the open Bay, low tide 74 billion m³ total, low tide	
Biological species:	>2700	
Usage:	ca. 1975	Estimated 2020
Recreation Activity, days	59,000,000	258,000,000
Commercial shipping, tons	160,000,000	300,000,000
Commercial fishing, kg	175,000,000	MSY may be exceeded for many species
Recreational fishing, kg	175,000,000?	
Water supply through systems, mgd	872	2320
Electricity generated, Gwh	68,000	882,000
Drainage:	Drainage area - sq. mi.	Mean annual flow, cfs
Susquehanna River (435 mi)	27,510	39,235
Potomac River (407 mi)	14,670	13,770
Rappahannock River (184 mi)	2,715	2,940
York River (130 mi)	2,660	2,660
James River (434 mi)	10,102	10,945
TOTAL BASIN	64,160*	76,890
Range in mean annual flow	49,000 (1965) - 131,800 (1972)	
Extreme low flow (week of Sept. 6, 1966)		4,720

***Exclusive of Bay and tributaries.**

each, comment on the general state of the Bay and of our present comprehension of it as a system, identify critical research needs for achievement of adequate protection and enhancement of the uses that are desired of this estuarine system, and, fortunately, describe several innovative and important improvements that have occurred during the past decade.

It is relevant to note that the U.S. Environmental Protection Agency has completed approximately three-fourths of a 5-year, $25 million Chesapeake Bay Program sponsored by Senator Charles McC. Mathias of Maryland. Principal emphasis is on toxic materials, submerged aquatic vegetation, excessive enrichment, and on improved management of water quality. About 60 principal investigators at 30 agencies and institutions are involved in about 45 projects. Only a Summary of Projects (Wells et al., 1979) and one technical report have been published at this time, although many preliminary data reports are available. Relevant studies will be noted throughout this report and identified as parts of the "Chesapeake Bay Program."


Chesapeake Bay Region
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SUSQUEHANNA
C&D Canal
N.J.
DELAWARE BAY
Baltimore
Washington
CHOPTANK
CHESAPEAKE BAY
PATUXENT
POTOMAC
RAPPAHANNOCK
YORK
JAMES
MD
VA
ATLANTIC
Cape Charles
Cape Henry
SCALE
NAUTICAL MILES
0 5 10 15 20 25
0 5 10 15 20 25
STATUTE MILES

Figure 1. The Chesapeake Bay.

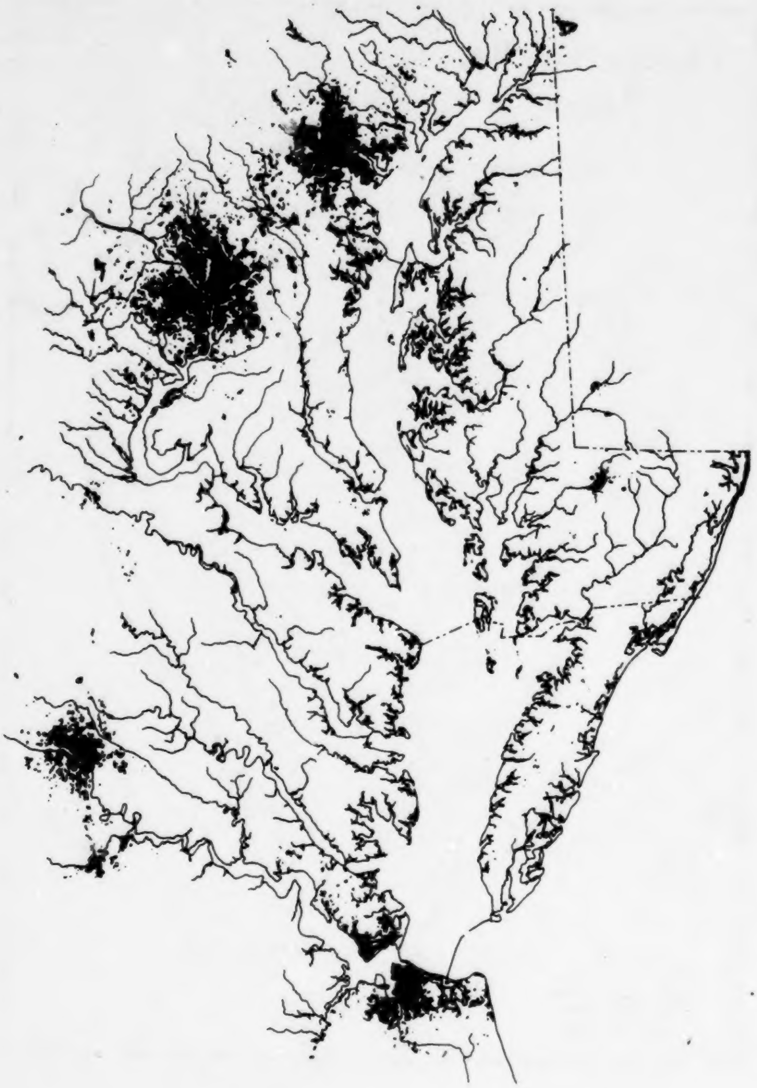

Figure 2. Chesapeake Bay region, indicating land used for residential and commercial activities as of 1973 (Corps of Engineers, Baltimore District, 1977).

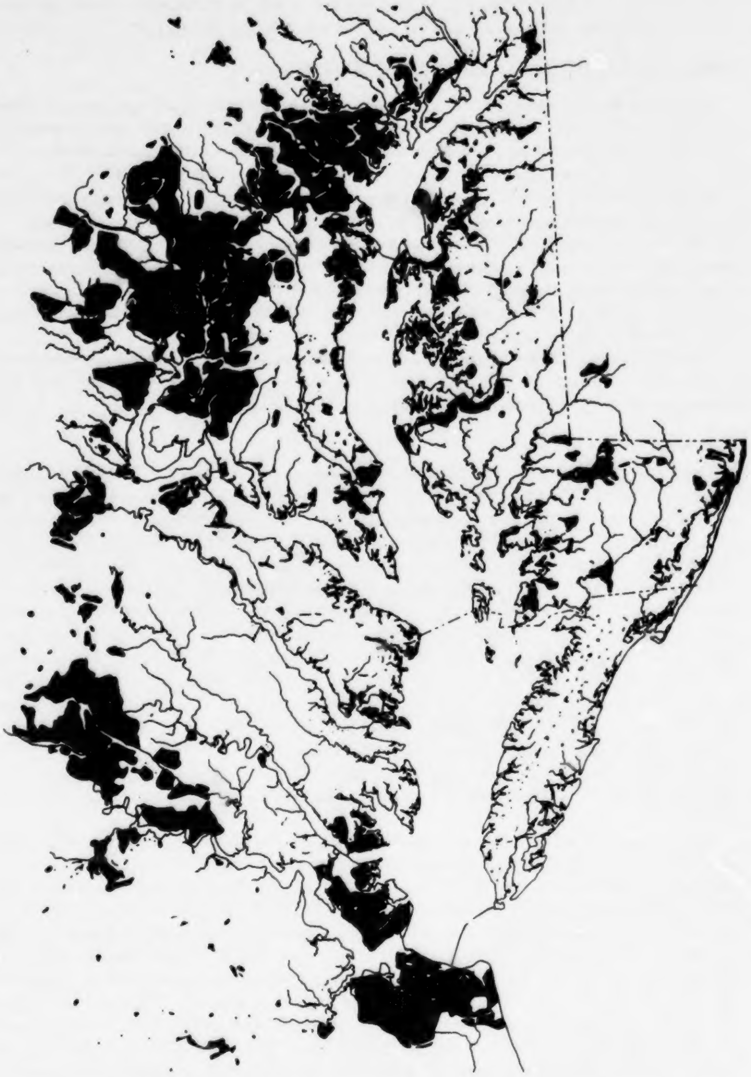

Figure 3. Chesapeake Bay region, indicating land used for residential and commercial activities as projected for 2020 (Corps of Engineers, Baltimore District, 1977).

On the basis of this review, a general assessment will be expressed on the question whether efforts to manage pollutants to this estuary are sufficient.

POLLUTION PROBLEMS AND PROGRESS

It is impossible to quantify the total set of pollutants reaching a large estuary under intensive use. It is difficult even to summarize them in general terms, with reasonable assurance that all anthropogenic materials and conditions injurious to present and potential uses of the system, i.e., all of the pollutants, have been included. A study of potential industrial chemicals, based on permits issued and on combined knowledge of chemical engineers, indicates that at least 545 compounds must be "disposed" of (GCA, 1979). In this case, there is good evidence of the input of hundreds or perhaps thousands of chemical compounds, plus pathogenic organisms, sediments, solid matter, color, heat, radioactivity, and materials that place demands on the oxygen supply in the Bay. The principal detrimental introductions merit comment, with emphasis on progress achieved and problems remaining.

Polluting materials and conditions are not introduced in neat and discrete categories. The content of the following subsections necessarily overlaps, but the groupings may be helpful in reviewing a tangle of issues.

Sewage

The scale of pollution problems from sewage is determined by the size of human population and related industrial activity, the composition of material entering sewage treatment plants, the types and degree of treatment, the quantity and pattern of release of liquid effluents, the placement of solid residue, and the total character of the receiving waters. All of these factors vary throughout the Chesapeake Bay system, but there is an increasing body of knowledge about magnitude and effects from these complex materials. The problems created by increasing release of partially treated sewage into the tributaries and main stream of the Chesapeake have long been recognized (Corps of Engineers, 1974; Corps of Engineers, 1977; Cronin, L., 1967; Ellis, 1973; Fish and Wildlife Service, 1970; McKewen, 1972; Schubel, 1972). They are concentrated near the metropolitan areas at Baltimore, Washington, Norfolk-Newport News-Hampton, and Richmond, but may appear near any population center, as in the Patuxent River where no city exists but where treatment plants concentrate regional wastes and add them to agricultural drainage. The multiple effects of domestic wastes on uses of the Bay are illustrated in Figure 4 (Ellis, 1973).

The magnitude and trends of sewage input are partially documented. The most comprehensive description and summary was produced in 1974 for all tidal waters of the Bay system (Brush, 1974). At that time, 35 plants using primary treatment and 207 employing secondary treatment released 945 million gallons per day (mgd) of treated wastes into the Bay that, when combined with 360 mgd from the Susquehanna, was 2.8 percent of the total freshwater input. The Corps of Engineers' Future Conditions Report projects that municipal wastewater treatment will increase (for the defined Bay region) from about 950 mgd in 1975 to about 1,770 mgd around the year 2000—an 85 percent increase (Corps of Engineers, 1977). Other numbers are available, but they are not comparable with these because only the report by Brush is based on both scanning of permits and direct observation of many of the plants.

The only major sewage treatment system providing advanced waste treatment is at Blue Plains, near Washington, D.C., with a design capacity of 309 mgd (Corps of Engineers, 1977). Advanced waste treatment is planned for several other sites, but review of costs against provable benefits is causing reassessments (Jensen, 1976).

In specific tributaries, estimates have been made of total loading. About 25 percent of the freshwater at the mouth of the James River is from sewage outfalls (Austin, 1979). For the Patuxent River, one estimate indicated that 74 percent of the low freshwater flow to the estuary in 1980 consisted of treated sewage. This is likely to increase to 84 percent by the year 2000 (Wilson, 1977).

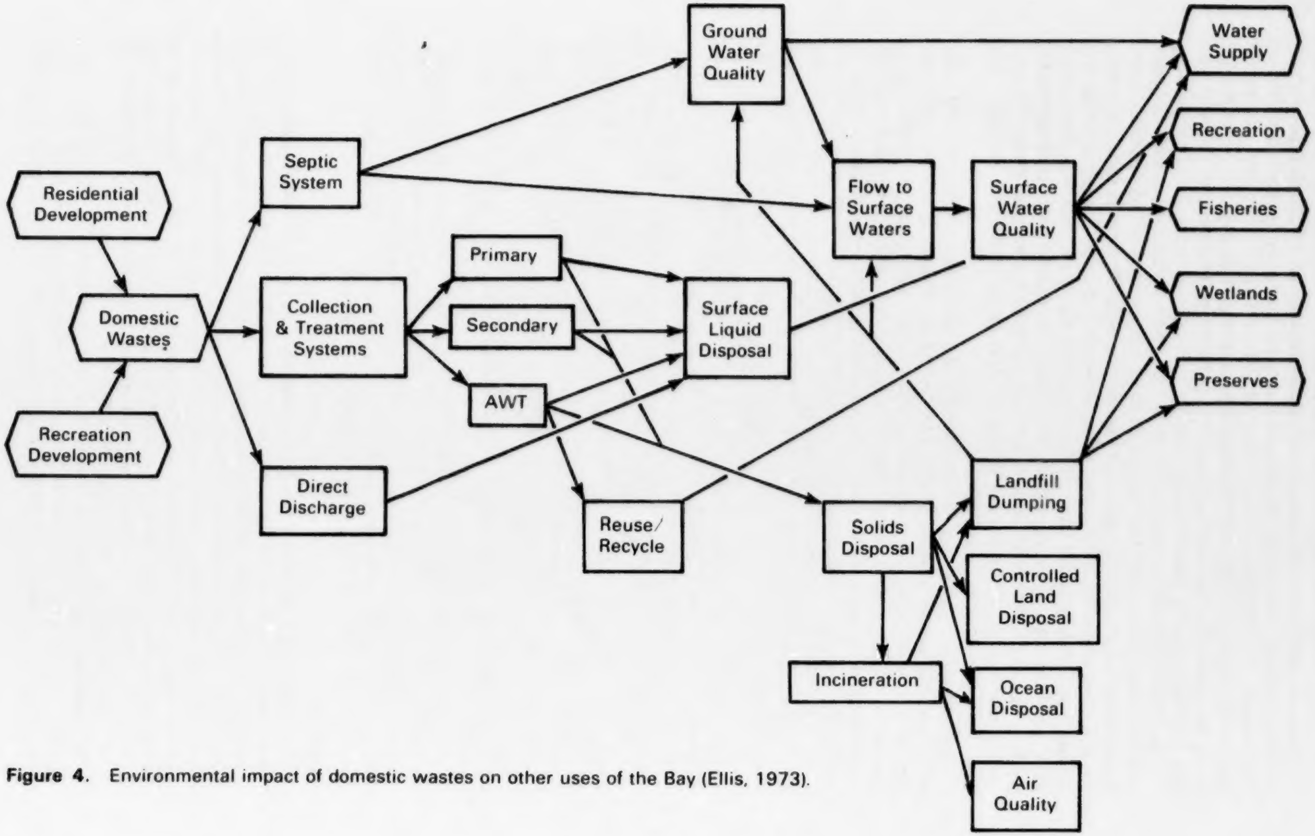
Residential Development
Domestic Wastes
Recreation Development
Septic System
Collection & Treatment Systems
Direct Discharge
Primary
Secondary
AWT
Reuse/ Recycle
Ground Water Quality
Surface Liquid Disposal
Flow to Surface Waters
Surface Water Quality
Solids Disposal
Incineration
Landfill Dumping
Controlled Land Disposal
Ocean Disposal
Air Quality
Water Supply
Recreation
Fisheries
Wetlands
Preserves

Figure 4. Environmental impact of domestic wastes on other uses of the Bay (Ellis, 1973).

The components of sewage effluents are, as always, diverse in response to inputs. Nutrients in many forms are present and so are trace metals, synthetic organic compounds, chlorine, detergents, particulate materials, and unnumbered other minor components. Washington, a residential and office-related city, produces at least 6 metric tons of phosphorus and 10 metric tons of nitrogen per day, but there are relatively few metals or other constituents of industrial origin (Schubel, 1972). Baltimore and the Norfolk area release larger relative quantities of nonnutrient chemicals.

The effects and management alternatives for some of these materials have been studied in the Chesapeake Bay region. Sewage sludges may not be deposited overboard in the system, and the potential alternatives of landfill, incineration, spraying for agricultural use, drying or composting with wood chips for land application, use as fuel, and dumping at sea have all been explored. Some are used, but no economically accepted and environmentally satisfactory method or set of methods for dealing with increasing quantities on a long-term basis has been found for the large population centers. The effects of sewage sludges on estuarine and freshwater fish have been summarized (Tsai, 1975).

Bacteria and viruses are introduced from septic tank overflow, overloaded treatment systems, and co-mingled sewage and surface water, usually during heavy rainfall. In the Chesapeake, the states maintain routine monitoring of shellfish beds, swimming beaches, and other sites by standardized coliform-based techniques supplemented by other testing. The area of shellfish beds closed because of excessive coliform counts has declined from about 385,000 acres in 1970 to about 240,000 acres in 1980; Maryland reduced its closure areas from 320,000 acres to 70,000 acres in that period (Garreis, 1980 and Wiley, 1980). Lowered numbers indicate both improvement in preventing contamination and reduction in the impact on the shellfish industry. The standard indicator techniques are recognized to be useful for some purposes, although both scientists and health officials recognize their substantial limitations (Colwell, 1977; Colwell and Kapper, 1978).

Recent research has partially documented the presence of potential human pathogens in the estuary, apparently from sewage sources (Colwell, 1977). Fecal coliforms can be very abundant, 24,000/100 ml of water, in Baltimore Harbor; *Salmonella* is frequently associated with fecal coliform organisms and is widespread, especially near cities. *Clostridium botulinum*, the agent of botulism, has been found at random sites over the Bay in sediments. Enteroviruses have received attention, but their significance in the Bay is not established. Many antibiotic-resistant bacteria have been observed in samples from shellfish and water. Fish have yielded preliminary evidence of contamination and possible infection by human pathogens near areas of sewer outfalls (Janssen and Meyers, 1968).

The principal impression from review is that, while efforts are made to protect human health, understanding of bacteria and viruses in the Chesapeake Bay is primitive and inadequate, both for those from sewage and those endemic to the system that play extremely important roles in the processes of the Bay (Colwell, 1977; Cronin et al., 1977). There are dangers in that situation.

Nutrients

The effects of nutrients on the Bay system and the costs of wise management of them present some of the most pervasive, complex, and perplexing of pollution problems. Nutrients arise from natural decay and runoff, from industrial activities, from disturbed land used for agriculture or development, and from sewage treatment plants. The present and potential effects of excessive enrichment have frequently been cited as a "major" problem or "the most serious" problem threatening the health and usefulness of the Bay system (Cronin, L., 1967; Cronin, L., et al., 1977; Federal Water Pollution Control Administration, 1969; McKewen, 1972; Perkinson et al., 1973; Pritchard, 1971; Schubel, 1972).

The national and global literature on nutrients is vast, and a large number of surveys, monitoring observations, research projects, and analyses have been completed in the Chesapeake Bay region (Champ, 1977; Corps of Engineers, 1974;

Corps of Engineers, 1977; Flemer and Heinle, 1974; Hydroscience, 1975; Jaworski, 1981; Jensen, 1976; Kuo et al., 1975; Laniak, 1979; Lippson and Lippson, 1979; Pheiffer et al., 1972; Roberts et al., 1975; Schubel, 1972; Sullivan et al., 1977; Tsai, 1975; Williamson, 1972). It would be inappropriate to detail all of these here; summary comments will be presented instead. Fortunately, much information has been synthesized in the last decade, so that highly useful analyses exist. These and several unusual specific cases will be reviewed.

An ambitious effort has been made to review all of the useful field data from the total Chesapeake Bay since 1913-1916, when the first were obtained (Heinle et al., in press). This review demonstrates the great difficulties inherent in such an effort. Records are lost; analytic techniques have been revolutionized; very few studies are sustained for decades; computerized sets are often incompatible, etc. It is possible to provide the following summary, however:

- The various portions of a large estuary differ fundamentally in components and processes related to nutrients and cannot be successfully lumped together under a single criterion or set of standards.
- The inherent variability of the estuarine system in response to rainfall, temperature, storms, and other factors vastly complicates interpretation of long-term data.
- Water quality in much of the total Chesapeake system has changed in recent decades because of increased nutrient enrichment.
- In the upper and middle Chesapeake Bay proper, and in several tributaries, nutrient content and algal concentrations have increased and light penetration has decreased.
- In some tributaries, available oxygen in deeper waters has been reduced, and annual variations in concentrations of oxygen are now more extreme.
- The lower Bay has been little affected, although the effects of enrichment have progressed down the main stem and tributaries over time.
- If concentration of Chlorophyll-*a* is used as the best available, albeit imperfect, gross indicator of threat from nutrient enrichment, the following guidelines can be stated for the Chesapeake Bay system:
    1) Estuarine waters of salinity less than 8 to 12 percent are moderately enriched if they have summer chlorophyll levels of 30 to 60 $\mu g/l^{-1}$; they are highly enriched, with potential of damage, if summer levels exceed 60 $\mu g/l^{-1}$.
    2) Waters of salinity higher than 8 to 12 percent are moderately enriched at 20 to 40 $\mu g/l^{-1}$ in summer, and highly enriched at levels above 40 $\mu g/l^{-1}$.
- The quality of algal populations may be as important as quantity, and each may at times be an objective in management.
- Reduction of nutrient input in the Potomac system has been demonstrated to be effective in improving the lower river and upper estuary.
- Either nitrogen or phosphorus may control production ratio in various areas of lower salinity, but nitrogen appears to dominate the mid- and lower Bay.
- The Bay has indeed been affected, but not yet to a critical degree except in some tributaries.
- Demographic projections and the present evidence of undesired effects are portents of possible serious future losses of quality and usefulness of parts of the Chesapeake Bay system.

A different, recent, brief summary has emphasized the very great role of sediments in the nutrient sequence in the Chesapeake Bay and other estuaries (Bertine et al., 1980).

Much of the nutrient entering the system becomes associated with the sediment and is deposited but may be regenerated from that sink at later dates. The processes involved are partially understood and include sediment deposition, biological activity in the sediment, resuspension, and interface transport. In the Chesapeake, the sediments return much of their nutrient burden to the aquatic systems, but a portion remains in the long-term sink.

Only limited and comparatively trivial efforts are yet underway to recapture nutrients and use them advantageously, despite the abundance of soil in the region impoverished by long cultivation of corn and tobacco.

Efforts to model and budget the nutrient sequence in the Chesapeake and its tributaries have been progressive and promise value in management. Among them:

- A two-dimensional quasi-linearized steady-state model of nutrient-phytoplankton interactions in the upper Bay was constructed to guide allocations (Hydroscience, Inc., 1975). It indicates that phosphorus is the controlling nutrient above the Potomac and that the primary nutrient sources are the Baltimore metropolitan area and the Susquehanna River. Much of the introduced nutrient is retained in sediments near the source.
- A one-dimensional tidal-time model was employed. Its purpose was to assist estimation of the environmental effects resulting from complete implementation of the Water Quality Control Act of 1972 and its amendments (Kuo et al., 1975).
- Applying model results to projection of biological impacts from changing the loading of nutrients and from oxygen-demanding pollutants yielded estimates. But this application also demonstrated the overwhelming complexity of such efforts (Roberts et al., 1975).
- From elementary raw data, a mass balance nutrient budget for the total Chesapeake Bay was constructed on a monthly basis (Laniak, 1979). Yearly loadings of nitrogen and phosphorus from point sources (the principal source of phosphorus), the tributaries (principal for nitrogen), and the air were estimated, as were outputs by advection and ocean loss. Seventy percent of the $1,212 \times 10^5$ kg/yr ($2,666 \times 10^5$ lb/yr) of total nitrogen received and 99 percent of the 164×10^5 kg/yr (360×10^5 lb/yr) of total phosphorus apparently remain in the Bay. Substantial seasonal variation occurs. The results are interpreted in relation to adequate monitoring of nutrient-related materials and processes.
- A major analysis of the sources of nutrients and the scale of "eutrophication" problems in estuaries included, among others, consideration of the external budget of the Chesapeake (Jaworski, in press). In 1971, 41,400 kg/day (91,080 lb/day) of phosphorus (69 percent from wastewater discharges) and 297,900 kg/day (655,380 lb/day) of nitrogen (68 percent from upper basin land runoff) entered the system. Variation between months and years, changes in the forms of nitrogen and phosphorus, and the gradients with distance downstream are discussed and related to a suggested scale of "eutrophication" and related to parts of the Bay and the total system.

From its inception, the Environmental Protection Agency's Chesapeake Bay Program has recognized and emphasized the importance of excessive nutrients in the Bay system. Under the inappropriate term "eutrophication" it has arranged for an important and valuable series of studies and research projects. None of the final reports are yet available, but the studies include (Wells et al., 1979 and Davies, 1980):

- Definition of Chesapeake Bay Problems of Excessive Enrichment or Eutrophication
- An Assessment on Nonpoint Source Discharge, Pequea Creek Basin, Lancaster County, Pennsylvania
- Evaluation of Water Quality Management Tools in the Chester River Basin
- Intensive Watershed Study (Patuxent River Basin)
- Evaluation of Management Tools in Two Chesapeake Bay Watersheds in Virginia
- Modeling Philosophy and Approach for Chesapeake Bay Program Watershed Studies
- Fall Line Monitoring of the Potomac, Susquehanna, and James Rivers
- Assessment of Nutrients from Various Sources

- Land Use and Point Source Nutrient Loading in the Chesapeake Bay Region
- Chesapeake Bay Circulation Model
- Water Quality Laboratory for Chesapeake Bay and Its Subestuaries at Hampton Institute
- Chesapeake Bay Nutrient Dynamics
- Chesapeake Bay Circulation and Water Quality Mathematical Models (two projects)
- Development of Assessment Tool to Evaluate Nutrient Transport and Fate in the Lower Susquehanna River
- Intensive Watershed Study (Chester River Basin)
- A Water Quality Modeling Study of the Chesapeake Bay Watersheds.

Growth in understanding and in the ability to manage nutrients rationally in the Chesapeake Bay region will be substantial. Since the population is predicted to double within about 40 years—with probable further subsequent increases—the need for adequate knowledge and effective management remains urgent.

Sediments

As in every estuary, there is continuous input of materials that become sediments. These arise from transport from the total basin by river water, from erosion of shores, from the products of biological activity, and, in some cases, from the sea. Sediments are deleterious to uses through filling of channels, progressive deposition in headwater areas, interference with light penetration, and smothering of benthic biota. Smothering is an occasionally serious event (Chesapeake Research Consortium, 1976), and reduced light penetration is under evaluation as a contributor to the extensive diminution of submerged aquatic vegetation over much of Chesapeake Bay since 1971 (Stevenson and Confer, 1978).

Inputs into the Chesapeake system are not precisely known, but the main stem of the Bay has been estimated to receive about 1.07×10^6 tons per year from the Susquehanna, $.60 \times 10^6$ tons from shoreline erosion, an unknown quantity from biological sources, and about $.20 \times 10^6$ tons from the ocean (Schubel and Carter, 1976). The large tributaries are sinks for their own materials, and part of the Bay load moves into them. The Bay is filling at an average rate of about .8 mm/year—about 3.5 to 4.0 mm/year at its head (Schubel and Carter, 1976). The reservoirs of the Susquehanna River dams were long considered to be sinks, but Tropical Storm Agnes flushed much of the accumulation in 3 days (Chesapeake Research Consortium, 1976).

The contribution of human activity to the input, dispersion, deposition, and resuspension and redeposition of sediments is not clear. In general, land clearing for agriculture or construction increases riverine input. Excessive nutrients increase plankton biological production, and wakes from ships and boats add to natural shoreline erosion. Particulate industrial wastes and urban runoff add sediments, and upland alterations that modify the "flashiness" of river flow will affect the sediment input and distribution. These have not been quantified for this estuary, but some of these human effects have been discussed (Schubel and Williams, 1976; Schubel and Wise, 1979).

The most important problems related to sediments are those associated with the continuous and external filling of channels where shipping or boating is desired and with the remarkable affinity of sediments for chemicals. Sediments have long been recognized as the source of large-scale economic and, more recently, environmental problems in the Chesapeake Bay region, where major cities (Baltimore, Washington, Richmond) lie on the fall line, above sites of natural deposition (Federal Water Pollution Control Administration, 1969). Channels must be wide and deep enough for shipping, and the beam, depth, and number of ships continue to increase (Villa et al., 1977). Maintenance of depths is a cost of pollution to the degree that the accumulating sediments result from human activities. In the Chesapeake, present channels require maintenance dredging of about 7,600,000 m^3/yr (10,000,000 yd^3/year). Proposed new channels and enlargements would generate about 197,600,000 m^3 (260,000,000 yd^3), including about 76,000,000 m^3 (100,000,000 yd^3) for the authorized completion of a 15 m (50 ft) channel from Baltimore to the Atlantic Ocean (McGarry, 1976; Villa et al., 1977). Although dredging can release suspended sediments and sometimes associated chemicals, the greatest associated problems are related to placement of the dredged materials. On-land sites are expensive and limited. The states of Virginia and Maryland as well as federal agencies object strenuously to wetland placement. Overboard placement is only conservatively permitted—and only for relatively clean materials, not defined as contaminated. And the season and dredging techniques are controlled. In Maryland, it is specifically illegal to place any of the sediments from Baltimore Harbor, containing large quantities of many pollutants as the result of centuries of casual use, overboard in the waters of Chesapeake Bay (Tsai et al.,1979). Long and acrimonious arguments have followed a proposal to create a large diked containment structure at Hart and Miller Islands to receive about 39,520,000 m^3 (52,000,000 yd^3) of contaminated sediments— and the project has not yet received full approval. Proposals to place contaminated sediments on old spoil sites and dewater them to produce useful land have not been enthusiastically received. Special concern has been expressed over the chemical characteristics of the water released. Meanwhile, Virginia has utilized a large diked area at Craney Island in the James River, but the capacity of that facility may soon be saturated (Villa et al., 1977).

Potential pollution from the dredging and placement of sediments has therefore become a principal issue in the Chesapeake Bay region, as in all coastal areas where major shipping occurs. This has been ranked as a major regional issue for the North and Mid-Atlantic regions (Horn et al., 1980). The input of sediment continues, but adequate accepted long-term solutions have not yet been found.

Sediments have, however, a different important relation to pollution in the Chesapeake Bay and other estuaries in that they are frequently associated, physically or chemically, with chemical materials introduced to or present in the estuary. Nutrients, heavy metals, pesticides, and many other inorganic and organic elements and compounds sorb to or chemically react with sedimentary particles. The source, routes, rates, and effects of these materials are largely determined by the related processes for the sediments.

The general components and processes for some sediment-related chemical materials are known for the Chesapeake Bay system, but new investigations are underway in the Chesapeake Bay Program (Office of Research and Development/EPA, 1980). These include intensive studies of the processes of nutrient deposition; nutrient modification in and release from sediments; research on the distribution, physical properties, budgets, and rates of sediments and sedimentation; research on the transport, fate, and transformation of metals related to suspended and deposited sediments; development of improved techniques for extraction and analysis of organic compounds in sediments and tissues; studies of the chemistry of pore waters; and examination of the relationships among sediments, associated chemicals, and the organisms living in the sediments. Only preliminary reports are presently available from these system-wide studies. They should make enormous contributions to the understanding of sediments and pollutants in the Chesapeake.

Flow Alteration

Modification of the flow into an estuary has not traditionally been considered as possible pollution, but it falls under a somewhat extended interpretation of introduction of deleterious conditions as the result of human activity. The many important influences of freshwater input upon estuaries were recently reviewed in an extensive literature summary (Snedaker et al., 1977) and were the subject of a major symposium (Coastal Ecosystem Project, 1980). Both demonstrate that substantial changes in the quantity or pattern of input can have enormous effects on the physical, chemical, and biological content and processes of an estuary—and therefore upon the human uses of it.

The Chesapeake was subjected to the effects of Tropical Storm Agnes in June of 1972. The storm, which had been of hurricane strength before it reached the Mid-Atlantic region, was obviously not a human activity, but it is relevant because it affected pollutants and revealed fundamental patterns of pollutant behavior and effects. Average basin rainfall over a 3-day period was in excess of 12 cm (5 in), with approximately one-third of the area receiving 30 cm (12 in) and isolated locations recording 46 cm (18 in) (Chesapeake Research Consortium, 1976). The effects of this 100- to 200-year storm event were dramatic, and observations were exceptionally thorough because the scientific community and, later, the management agencies recognized the importance of the event and arranged for extensive detailed observation. The Virginia Institute of Marine Science, the Chesapeake Biological Laboratory of the University of Maryland, and Chesapeake Bay Institute of the Johns Hopkins University were especially prompt and effective in their research, and the Corps of Engineers provided essential financial assistance. Highlights of observations include:

- The dominating Susquehanna had 7-day flows 15.5 times greater than normal, peaking at 1,130,000 cfs. The James flowed at as much as 24.4 times normal and the Potomac at 19.7 times normal.
- The Susquehanna debouched more sediment in 10 days than during the preceding 10, perhaps 25 or more, years, about 31 million metric tons against an annual average of one-half to one million tons.
- Dissolved nitrates and nitrite were 2 to 3 times normal in the northern half, but little affected downstream. Phosphate remained near normal. The nutrients were rapidly lost to the sediments.
- Trace metal and pesticide concentrations were not drastically changed. Oil input was substantial.
- Soft-shell clams, oysters, and some aquatic plants suffered heavy mortalities. Finfish, crabs, and hard clams were relatively unaffected.
- Bacterial contamination forced temporary closure of the Chesapeake Bay and its tributaries to the harvest of shellfish, but reopening was possible within weeks or a few months.
- The entire biological community was disrupted, but most effects had disappeared after 2 years.
- The Chesapeake Bay ecosystem demonstrated great resilience to this extreme natural event.
- The storm increased heterotrophic activity in parts of the Bay, reduced phytoplankton in the upper Bay but stimulated greater production in the lower Bay, raised nitrogen in the lower estuary, moderated algal production in some areas by shading, and was followed by reduction in dissolved oxygen concentration (Chesapeake Research Consortium, 1976).

The studies are detailed in the last-named reference by a large number of authors who cannot be individually credited here. They provided a remarkable record and achieved important advances in estuarine science.

An artificially constructed waterway, the Chesapeake and Delaware Canal, was dug in 1829, converted from a locked sequence to a sea-level canal in 1927, and enlarged from 8 by 75 m to 10 by 135 m (27 by 250 ft to 35 by 450 ft) in the period between 1958 and 1972. In 1974, nearly 11,000 vessels carrying 12,400,000 tons transited the canal. Concern over the possible environmental effects of enlargement, including possible diversion of large volumes at periods of low flow, resulted in extensive research and analysis of the hydrologic patterns created and of effects on the biota (Cronin, 1977). It was concluded that the physical hydrography, chemical environment, and biotic populations of the canal and areas of approach had been substantially altered. Long-term net transport from the Chesapeake was estimated to increase from 900 to about 2,450 cfs in a highly complicated hydrologic sequence with eastward and westward maximum flows of about 48,800 and 37,900 cfs. A new site for intensive spawning of striped bass was created by building the sea-level canal, but it may transport eggs and larvae into unfavorable water. Biota are generally diverse and abundant. Later analysis has shown that longer periods of hydrographic observations than those of this study are required for accurate estimation of the net transport in this very dynamic pipeline between estuaries. The best present estimate is that there is probably little long-term net transport despite massive short-term movement in response to tides and meteorological events (Pritchard, 1980).

Future modifications in freshwater flow are of serious concern. Demand for consumptive loss from the principal tributaries for irrigation, consumptive industrial uses, and modest export for water supply outside of the Chesapeake watershed may reach 5,360 cfs in summer by 2020 (Robinson, 1980). In contrast, the low flow of record, experienced in 1966, was 4,720 cfs and the long-term average for the driest month, September, is 28,400 cfs. The Baltimore District of the Corps of Engineers has initiated extensive studies involving the Chesapeake Bay Hydraulic Model and contracted biological analysis to estimate the changes in salinity that might result from future losses and the effects on specific biota and on the uses of the Bay system (Shea et al., 1980; Withers, 1979). At the time of writing, the effects have been simulated in the huge model, the largest estuarine hydraulic model in the world, but analysis has not been completed.

Heat

Heat, in the form of wasted energy from large power plants fired by fossil fuels or nuclear energy, caused some of the most vociferous arguments in the Chesapeake Bay region in the late 1960s and through the 1970s. They focused on the proposals to construct nuclear facilities at Calvert Cliffs near Cove Point and at Douglas Point in the center of the striped bass spawning areas of the Potomac River. Concern centered on effects of heat on migratory aquatic species and on entrainment losses from the combined impacts of mechanical, thermal, and chemical stresses.

It is not appropriate to detail here the long and convoluted efforts to achieve adequate environmental protection along with adequate supplies of electricity. Several principal events and trends have emerged:

- High temperatures have been precluded by general restriction of thermal rise across condenser systems to 10°F. This has, however, required enormous quantities of water—Calvert Cliffs requires an estimated 5,500 cfs of Bay water, making it the fourth largest "tributary" of the Bay.
- Large areas are warmed. Calvert Cliffs warms about 500 square miles of water less than 10°F as heat moves through the water to the atmosphere. Against high natural variation in termperature, any effects are difficult to identify and evaluate.
- More recent permits have required cooling towers rather than pass-through cooling, precluding heating effects on the open system and reducing, but not eliminating, aquatic losses.

Maryland has had a Power Plant Siting Law since 1971, which provides a tax of .1-.3 mil per kw of production. The resultant fund of about $5 to $6 million per year has been applied in a wide variety of research projects related to generating operations and their effects as well as in evaluation of proposed sites, monitoring, and acquisition of potential sites for utilities. About 290 research and study reports have been supported, most of which deal with estuarine questions. The Second Thermal Workshop of the U.S. International Biological Program was held at the Chesapeake Biological Laboratory of the University of Maryland and focused on research in the Chesapeake Bay and other estuaries (Mihursky and Pearce, 1969). Twenty-nine papers and workshop summaries were presented. It is not yet possible to determine whether or not the management of generating plants has eliminated significant injury to estuarine uses, but it is clearly based on a large and expanding body of relevant knowledge.

Future projections suggest increase in "demand" for electricity for the Chesapeake Bay Market Area of approximately 13.5 times from about 1975 to 2020 (Corps of Engineers, 1977). While this projection may be modified by changes in priorities and costs for fuel, it is clear that effective protection of water quality must indeed be based upon well-informed and careful management of the facilities.

Spills

Accidental spills and deliberate releases are never fully documented at large centers of shipping and industry like Baltimore and Norfolk-Newport News, but the recording and response to these localized accidental releases and to accidents in transit are improving under 1970 federal direction to the U.S. Coast Guard and improved cleanup programs by Virginia, Maryland, and involved industries. Petroleum products and toxic chemicals cause the greatest concern, but hundreds of other materials are sometimes released into the system. The general topic of "Prevention and Control of Spills" was treated extensively by a workshop report and extended discussion at the Bi-State Conference on the Chesapeake Bay in 1977 (Hess et al., 1977). In 1975 and 1976 an annual average of 740 spills releasing 334,700 gallons of materials were reported. Petroleum products provided 72 percent, mostly heavy oils. Several serious groundings and other accidents have occurred, and the workshop report and others at that Conference stressed the critical importance of adequate operating requirements for vessels, safe techniques for transfers, improved vessel traffic management, better data management, and increased public concern and action (Hess et al., 1977; Villa et al., 1977).

Petroleum products have been the center of increased attention. Bulk oil traffic, about 39,000,000 short tons in 1970, has been projected to double by the year 2020 (Corps of Engineers, 1977). Spills of up to 240,000 gallons in this nearly enclosed, slowly flushed, biologically useful system have prompted much concern. Drawing from research and experience both within the Chesapeake region and from other sources, several summaries and general analyses of probabilities, fate, and effects of oil spills have been developed. (Chesapeake Bay Foundation, 1977; Cronin, L., 1976; Farrington, 1977; Hess et al., 1977; Rose, 1974). At least one reviewer noted that oil spills in the Chesapeake are far below a reported world average of .16 percent of total transport and attributed this fact to care in navigation, piloting, handling in port, and other effective methods of prevention (Cronin, 1976).

The fate of a hypothetical oil spill near the center of the Chesapeake Bay was modeled with estimation of the sites, kinds, and magnitudes of effects (Kelly, 1976). A 120,000-gallon crude oil spill might contaminate 144 to 320 km (90 to 200 mi) of shoreline and substantially damage wetlands, waterfront, and commercially valuable invertebrates over at least 2 to 4 years. The Bay appears to be on borrowed time, and a major spill and very serious damage seem inevitable within a decade or so (Cronin, 1976; Chesapeake Bay Foundation, 1977).

Only one oil refinery is operated in the Bay region, at Yorktown, Virginia. No catastrophic effects have been reported, but there has been strong opposition to a refinery proposed in the Baltimore area, a refinery at Portsmouth, Virginia, and a large oil terminal at Piney Point in the Potomac River. Objections have been based on the dangers of spillage from transfers and operations, the existence of critically important aquatic resources near each of these sites, and the probability that deleterious effects would last for many years. The Baltimore and Potomac proposals have been defeated or withdrawn, but the Portsmouth terminal and refinery has, at this time, passed many local, state, and federal hurdles. It has not yet been constructed (Chambers, 1979).

Toxicants

Metals and other chemicals that can be detrimental to uses of the Bay are introduced in sewage and also from industry, accidents, surface runoff, and from the tributary rivers. One study found that sewage treatment plants introduce about as much cadmium, copper, zinc, and lead to the Bay system as is received from the tributary rivers (Huggett et al., 1974). For manganese, iron, cobalt, and nickel, the river inputs substantially exceed treatment plants. Near one large sewage outfall, concentrations of heavy metals in sediments were 10 to 100 times those in uncontaminated areas, indicating that most of the metals were deposited near the source. Other surveys and studies of heavy metals could not identify the sources, especially in Baltimore Harbor and Elizabeth River, where large quantities of metals are present in sediments, but both industries and sewage treatment plants have contributed (Cronin et al., 1974).

Potentially toxic chemicals are frequently, perhaps continuously, introduced into the Chesapeake Bay from sources other than sewage treatment plants. They have been identified as one of the three most serious threats to the health of the Bay (Huggett et al., 1977; Cronin et al., 1977). Substantial efforts have been made to preclude introduction of toxicants, as in the Federal Toxic Substances Control Act of 1976, Maryland's Safe Disposal of Designated Hazardous Substance Act, and similar legislation in Virginia (Huggett et al., 1977). While it is by no means certain that industrial and domestic wastes meet present standards of National Pollution Discharge Elimination System permit statements, every *new* industry is required to assure compliance. Principal problems appear to arise from old industries, old sewage treatment systems, and the vast accumulations of metals, oils, and unidentified pollutants in the sediments of Baltimore Harbor, Elizabeth River, and, to a lesser concentration, other sites (Jaworski, 1981; Office of Water Planning and Standards, 1977; Tsai et al., 1979).

The Environmental Protection Agency's Chesapeake Bay Program gave early and high priority to some of the problems of toxics in the food chain. They have supported or are supporting projects on:

- Sedimentology of the Chesapeake Bay
- Baseline Sediment Studies to Determine Distribution, Physical Properties, Sedimentation Budgets, and Rates
- Chesapeake Bay Sediment Trace Metals
- The Characterization of the Chesapeake Bay: A Systematic Analysis of Toxic Trace Elements
- Investigation of Organic Pollutants in the Chesapeake Bay
- Interstitial Water Chemistry
- Sediment and Pore Water Chemistry
- Monitoring Particle-Associated Toxic Substances and Suspended Sediment in the Chesapeake Bay
- Fate, Transport, and Transformation of Toxics: Significance of Suspended Sediment and Fluid Mud
- Animal Sediment Relationship
- The Biogenic Structure of Chesapeake Bay Sediments
- Inventory and Toxicity Prioritization of Industrial Facilities Discharging into the Chesapeake Bay Basin
- Chemistry of Wet and Dry Fall to Lower Chesapeake Bay
- Aqueous Effluent Concentrations for Biotesting
- Toxic Point Source Assessment of Industrial Discharges to the Chesapeake Bay Basin
- Biofractionation of Industrial Discharges
- Evaluation of Bioassay Methodology for Application to Chesapeake Bay and Other Estuaries (Davies, 1980; Office of Research and Development, 1980).

Excellent descriptions of the chemical burden of waters, sediment, and biota will result, and much is being learned about the sources. Only the last project is directed toward improved comprehension of the biological effects of toxicants in this and other estuaries—a critical area for future studies.

Chlorine is the most widely used biocide to disinfect the effluents from sewage treatment plants, some food processing plants, and other materials. It is also employed to minimize sliming and fouling in the tubes and pipes of generating stations for production of electricity. Recent figures in Maryland indicate that chlorine release to Bay water (assuming no degradation) is about 12 million kg/yr (27 million lb/yr) from sewage treatment plants and 1 million kg/yr (2.2 million lb/yr) from power generation (Davis and Middaugh, 1977). Perhaps 1 percent of this becomes halogenated organic compounds and persists in the system. The toxicity of chlorine and of chlorine-produced oxidants has been established for some Chesapeake species, and the larval stages are generally the most susceptible (Bertine et al., 1980; Chesapeake Research Consortium, 1977; Davis and Middaugh, 1977; Roberts et al., 1979).

Massive kills of four species of fish in the James River in 1973 resulted in vigorous cooperative studies and analyses by state agencies (Douglas, 1979; Virginia Marine Resources Commission, 1979). Chlorine and its derivatives were clearly implicated. Operational improvements in treatment plants, dechlorination, and perhaps the use of bromine chloride, an effective disinfectant of lower estuarine toxicity, are useful in reducing mortalities—which were in fact lowered to acceptable levels (LeBlanc et al., 1978; Douglas, 1979). A chlorination workshop in 1977 provided 16 summaries of available knowledge of the fate and effects of chlorine, the problems and techniques involved in analysis of chlorine and residual chemicals, uses in cooling systems, bioassay of plants and animals, and the behavioral and physiological responses of estuarine organisms (Block and Helz, 1977).

Recently, concern for chlorine effects has again surfaced, and controversy over the balancing of protection of public health versus injury to valuable estuarine species is receiving fresh attention (Horton, 1980). A public conference titled "Chlorine—Bane or Benefit?" is scheduled for the spring of 1981.

Herbicides have been used for agricultural purposes, especially in no-till practices, in increasing quantities in the last decade. No-till practices, which reduce runoff but require the use of herbicides, began about 1969, and by 1977 application of triazines in the Bay region reached 1,500 to 15,000 tons, depending upon the estimator (Citizens' Program for the Chesapeake Bay, 1978). A small percentage, on the order of 1.5 to 2.0 percent, may be carried off into water. Some reaches the estuary. In the same period, submerged aquatic vegetation progressively declined in abundance over much of the Chesapeake system—about 50 percent in the number of sites (among 625) that were vegetated in Maryland, and extensively in Virginia (Citizens' Program for the Chesapeake Bay, 1978). The coincidence has been noted and argued extensively (Cronin et al., 1977). An extensive summary of knowledge about such vegetation in the Bay noted that these chemicals can injure such plants in laboratory experiments, that the extent of damage in the Bay system is unknown, and that a considerable number of other factors may affect submerged aquatic vegetation (turbidity, salinity, fauna, temperature, sediments, chlorine, nutrients, boating, etc.) (Stevenson and Confer, 1978). The Chesapeake Bay Program includes related projects on (Wells et al., 1979; Davis, 1980):

- Distribution of Submerged Vascular Plants in the Chesapeake Bay, Maryland—1978 and 1979
- Distribution and Abundance of SAV in the Lower Chesapeake Bay, Virginia—1978 and 1979
- *Zostera marina:* Biology, Preparation, and Impact of Herbicides
- Submerged Aquatic Vegetation in the Chesapeake Bay: Its Role in the Bay Ecosystem and Factors Leading to Its Decline
- Assessment of the Potential Impact of Industrial Effluents on Submerged Aquatic Vegetation
- Effects of Recreational Boating, Turbidity, and Sedimentation Rates in Relationship to Submerged Aquatic Vegetation
- Factors Affecting and Importance of Submerged Aquatic Vegetation in Chesapeake Bay (Wells et al., 1979; Davis, 1980).

There are also valuable projects on the functional roles of aquatic vegetation and their use as habitats for important species.

It is to be hoped that the important uncertainties about possible relationships between valuable agricultural activities and valuable estuarine resources will be definitively resolved and either corrected or dismissed, as may be appropriate.

Chlorinated hydrocarbons were the subject of both concern and research, and the knowledge of their effects was brought before Chesapeake Bay interests (Walsh, 1972). Concern has declined along with use, although decay products have been reported in Bay sediments. Special problems have arisen with the storage of PCBs in inadequate facilities, the transfer to improved containment, and proposed incineration, but no serious pollution is known to have occurred.

Radioactivity has not been shown to be detrimental to the uses of the Chesapeake Bay. There are two nuclear generating stations on the estuary. The well-known Three Mile Island is in the middle of the largest source of freshwater. Completion of a series of nuclear plants is underway with some in the series operative and some proposed on the Susquehanna. Cleanup and decontamination of Three Mile Island have not been completed, and 2,646,000 l (700,000 gallons) of highly contaminated water must be disposed of. Public agencies and the citizens of the Chesapeake area are deeply concerned but hopeful that the established safety levels for radionuclides are valid.

The Kepone Saga

Release into the James River of a relatively unknown pesticide developed to control ants, cockroaches, and Central American banana root borers created an estuarine catastrophe that will last indefinitely. The later stages of the sequence are well documented and may be summarized as follows (Associated Press, 1980; Cronin et al., 1979; Huggett and Bender, 1980; Huggett et al., 1980; Lunsford et al., 1980; Nichols et al., 1979):

- From 1966 to 1975, Kepone was discharged into the environment, the sewage treatment system, and the tidal river at Hopewell, Virginia, on the James River.
- Recognition of dangers occurred when employees displayed serious health problems and subsequent investigation uncovered heavy contamination of soils, water, and estuarine sediments, and threatening quantities in benthic and pelagic organisms.
- Kepone is toxic to many aquatic species, concentrated in a number of species, and transferred through the food web.
- About 140 kg (308 lb) of Kepone are concentrated in the biota of the system.
- An estimated .5 metric tons now remain in or near the source area. 10.4 metric tons are distributed in the sediments of the estuary over a distance of 88 km (55 mi).
- All major components of the James estuary contain Kepone—biota, water, and sediments. The pathways of cycling have been approximated and include plants, benthos, plankton, nekton, and birds.
- Kepone is highly persistent, and a wide variety of proposed corrective measures (stabilizations in sediment, covering of sediments, incineration, etc.) are costly and unfeasible. The least expensive plan would cost over $3 billion.
- The economic impact is enormous, since the river was closed in 1976 to the entire valuable recreational and commercial fisheries.
- Modest improvement has occurred, and recreational fisheries are now permitted to retain their catch and short-exposure fish such as shad can be retained. Medical tests are reported to show that humans can eliminate Kepone more effectively than test rats and mice.
- Kepone is slowly buried by more recent sediments in areas of high sedimentation, but high contamination persists in much of the biota.
- Reexamination of old analytic data and archived samples revealed the early history of introduction when the problem was unsuspected (Cronin et al., 1979).

Documentation of the fate, effects, costs, and possible remedial measures has been exceptionally thorough. Involved investigators have noted that "our ability to correct such widespread contamination is extremely limited both technically and economically" (Huggett and Bender, in press) and that "Kepone is an example of but one of thousands of potentially toxic new substances being manufactured every year" (Nichols et al., 1979).

Ultimately, recommendations for dealing with Kepone rest with the U.S. Environmental Protection Agency, working in coordination with Virginia and Maryland. This is a massive burden to have resulted from inadequacy in operation, monitoring, and regulation.

RECOGNITION OF THE CHESAPEAKE BAY AS A SYSTEM

With increasing emphasis, the Chesapeake Bay and its tidal tributaries are being regarded, studied, and managed as a single entity with physical, chemical, and biological continuity. In an estuarine system of this size, diversity, political subdivision, and complexity, the approach has been achieved slowly and despite parochial reluctance. The earliest recognition of the Bay's unity came to those navigators who used the great single transportation network it provided. Subsequently, the scientific community studied the physical system, the migratory species of invertebrates, fish, and birds, the chemical continuity, and other aspects that required consideration of its totality (Chesapeake Research Consortium, 1976; Cronin, L., et al., 1971; Cronin, W., 1971; Huggett et al., 1977; Kuo et al., 1975; Lynch et al., 1977; McErlean et al., 1972; Schubel, 1972). There is now broad but incomplete acceptance that "an ecosystem must be ordered and husbanded within its own terms" (Hedgepeth, 1972), and that the total Chesapeake is indeed such an ecosystem.

Approach to the entity has been demonstrated in several areas in the last decade. A series of conferences, supported by the states, citizens' groups, professional societies, federal agencies, and several coalitions of state and federal agencies all focused on the total Chesapeake Bay system (American Water Resources Association, 1976; Bergoffen, 1971; Chesapeake Research Consortium, 1977; National Aeronautic and Space Administration, 1972; National Aeronautic and Space Administration, 1978; State of Maryland, 1968; Washington Academy of Sciences, 1972). The reports from these contain valuable overviews and integrated summaries.

Research analysis and planning have become more comprehensive. The Chesapeake Research Consortium, Corps of Engineers, and Environmental Protection Agency have developed broad program statements and implemented them (Beers et al., 1971; Office of Research and Development, 1980; Prentiss, 1972). The design of research for some of the species considers the full estuary (State of Maryland and Commonwealth of Virginia, 1980). Analysis of important issues and of the application of research in their resolution has employed the complete Bay region as the target (Douglas, 1979). Total system models have been attempted for nutrients (Hydroscience, Inc., 1975; Jaworski, 1981; Kuo et al., 1975; Laniak, 1979) and one conceptual ecological model for Chesapeake Bay has been prepared (Green, 1978). The largest estuarine hydraulic model, a fixed-bed geometrically distorted model at 1:1000 horizontal scale and 1:100 vertical scale, has been constructed, verified, and employed in an initial series of studies (McKay, 1976). The model occupies about 9 acres on Kent Island. Numerical analogues for hydrographic behavior and containment dispersion in one-, two-, and three-dimensional models exist for various portions of the Bay and for the system (Ulanowicz, 1976). Further development and application of two- and three-dimensional models are incorporated in the Chesapeake Bay Program. Models are essential for theoretical and practical purposes, and these values are being employed and will be enhanced.

Data systems abound. Twelve institutional systems have been identified (Kohlenstein, 1972), and a large number of files exist for specific purposes. Quality control varies as well as purpose and scope, and no unifying solution has yet been effected. The most optimistic possibility for unification appears to be in a Primary Chesapeake Bay Data Bank, employing common and accepted data units, terms, and programs, plus local or institutional specialized data systems as compatible as possible with each other and with the Primary Bank (Cronin et at., 1979). This has not been achieved and is not in sight.

The most recent effort to unify consideration and study of Chesapeake Bay is the "Chesapeake Bay Research Coordination Act of 1980" introduced by Senator C. McC. Mathias and Representative Robert Bauman, both of Maryland. It is intended to assure effective research planning and coordination of all Bay-related research supported by federal funds (Congress of the United States, 1980). The law did not go into effect until October 1981.

Summaries and syntheses are increasingly available. In addition to, or as part of, the conference reports noted above:

- Individual authors have presented overviews of the physical, chemical, and biological knowledge of the Bay and commented on its condition (Cronin, L., 1967; Cronin, L., 1978; Cronin et al., 1977; Lippson and Lippson, 1979; Pheiffer et al., 1972; Schubel, 1972).
- Two atlases have been completed, presenting general data on the biota and environment of the Maryland portion of the Bay and detailed information on the Potomac River estuary (Lippson, 1973; Lippson et al., 1980).
- Teams have developed concensus summaries, interpretations, and recommendations on various problem areas and on the "condition" of the Bay (Cronin et al., 1977; Federal Water Pollution Control Administration, 1969; Fish and Wildlife Service, 1970; Hess et al., 1977; Hugget et al., 1977; Lynch et al., 1977; Sullivan et al., 1977).
- Historical trends have been examined (Heinle et al., in press).
- One valiant description has been completed of the present (ca. 1973) and future (2000 and 2020) conditions of the Bay "including, but not limited to the following: navigation, fisheries, flood control, noxious weeds, water pollution, water quality control, beach erosion, and recreation" (Corps of Engineers, Baltimore District, 1974, 1977).

Federal law treats the Chesapeake Bay as a unit, but administrative practices do not always do so. The tidal Chesapeake system is in one region (III) for many federal agencies, divided between two in several, and split into three districts of the Corps of Engineers (Friedlander, 1979). The State of Maryland and the Commonwealth of Virginia know well where the boundaries are, but each frequently refers to its portion of the system as "the Chesapeake Bay" (Wallace et al., 1972).

IMPROVED MANAGEMENT OF POLLUTION

Since implementation of The Federal Water Pollution Control Act of 1972 (PL 92-500) and its amendments, major changes have occurred in the management of pollutants in and around Chesapeake Bay. Maryland and Virginia are "designated" states, with programs accepted by the U.S. Environmental Protection Agency, and therefore hold primary responsibilities for water quality. Virginia created a State Water Control Board in 1972 that has expanded substantially and has vigorously attacked Bay problems. Maryland has undergone several administration changes, and primary responsibility for water quality was recently placed in a new Office of Environmental Programs in the Department of Health and Mental Hygiene, which is also responsible for air quality, hazardous substances, and waste management. Both states are negotiating agreements with the U.S. Environmental Protection Agency for more efficient planning, management, and technical assistance. Efficient meshing of federal and state responsibilities, action, and funding is difficult, but it is being seriously attempted.

In the last decade, the federal government has made three major attempts to improve understanding and management of water quality in the Chesapeake region. The Corps of Engineers' Chesapeake Bay Study, authorized in 1965, assembled its

Existing Conditions Report in 1974 and Future Conditions Report in 1977, supported three years of hydrographic study of the Bay, constructed the Chesapeake Bay Model, and began its use with outside support for some of the studies (Corps of Engineers, Baltimore District, 1974, 1977; McKay, 1976; Shea et al., 1980; Withers, 1979). The National Science Foundation, through its program on Research Applied to National Needs (RANN), funded the operation of the Chesapeake Research Consortium, established in 1972 by The Johns Hopkins University, University of Maryland, Smithsonian Institution, and Virginia Institute of Marine Science to conduct interdisciplinary and interinstitutional research on principal problems, and supported that effort over a period of about 4 years. In fiscal year 1976, Congress directed the U.S. Environmental Protection Agency to conduct a 5-year, $25 million Chesapeake Bay Program of research directed to assess the principal adverse factors impacting the Bay's environment, improve related data systems, and assist in better management of the system (Office of Research and Development/EPA, 1980). The products of the RANN investment are completed and widely disseminated; the Corps' reports are readily available, but the model studies have not been completed; and the results of the U.S. Environmental Protection Agency's program are beginning to appear. Such uneven federal funding of relatively short-term attention to long-term problems has important value, but is inadequate in meeting the needs of the region (Cronin, 1979).

Other important federal and state-federal programs were also initiated. Both Virginia and Maryland established Sea Grant Programs and Coastal Zone Management Programs in cooperation with the National Oceanic and Atmospheric Administration. Sea Grant Programs appear to be well established. Maryland has established a Coastal Resources Division as a permanent center for coastal zone efforts, while Virginia has chosen to forego federal support and leave these matters as tasks of various agencies.

Research efforts have escalated, albeit unevenly and largely in response to funding opportunities. The Chesapeake Research Consortium functions through the four largest academic centers, with about two-thirds of the non-federal scientists of the area and the principal research facilities. It exists to identify the problems of the region, to conduct multi-institutional research toward their solution, and to assist management agencies. The Smithsonian Institution has established a Chesapeake Bay Center for Environmental Studies, and the University of Maryland has regrouped its Bay-related programs to create a Center for Environmental and Estuarine Studies as a new branch of the University. All of these and a large number of other federal laboratories, academic institutions, consulting firms, and other organizations have participated in research related to pollution and the Bay ecosystem (Chesapeake Research Consortium, 1978).

The states of Maryland and Virginia have undertaken cooperative ventures that are highly innovative in this region where state rights are rigorously protected. As the result of a study by the Chesapeake Bay Legislative Advisory Commission, created by legislative action in 1978, a permanent Chesapeake Bay Commission was established in 1980 by the two states. Present roles are limited to review of long-term needs, advisement of state legislative and executive branches and the federal government, and assessment of coordinated efforts between the states and with the federal government (Gartlan et al., 1980). In 1979, a complementary Bi-State Working Committee was established for continuing direct interaction between the executive agencies of Virginia and Maryland. The Commission and Committee may be highly important in assessment and control of pollution.

It is appropriate to note that a Susquehanna River Basin Commission and Interstate Commission on the Potomac River exist, so that there are now complementary commissions for the entire watershed. The managerial network would appear to be in place. The achievements obviously lie in the future and cannot now be assessed.

PUBLICITY, EDUCATION, PUBLIC PARTICIPATION, AND PERCEPTIONS

Notable improvements have occurred in the availability of reliable information about Chesapeake Bay, media attention, and in the involvement of its many publics in the processes of management. *The Chesapeake Bay in Maryland—an Atlas of Natural Resources*, presenting extensive material in readable narration and excellent graphics, is widely used in schools as well as by scientists and managers (Lippson, 1973). An *Environmental Atlas of the Potomac Estuary* contains similar graphics, detailed summary of many aspects of this sub-estuary, text-like presentations on the physical, chemical, and especially the biological aspects of estuaries in rich detail, and a set of folio maps on cultural landmarks, topography, sediments, aquatic vegetation, benthos, spawning region migrations, wastewater, and boat waste discharges (Lippson et al., 1980). This is the finest estuarine treatment of this type. No single complete description and interpretation of Chesapeake Bay has yet been achieved, but interest has been expressed.

Movies, slide shows, booklets, and other formal educational materials are increasingly available, as are newsletters, brochures, and periodic information from citizens' groups, agencies, and commissions. Some school systems have units on Chesapeake Bay. Frequent press attention is given to news and views about the Bay, and major features appear from time to time in local and regional papers, on television, and in the National Geographic and other magazines (Fisher, 1980; Hoffman, 1979; Kanigel, 1979; Perkinson et al., 1973). These usually deal with specific problems (Kepone, nuclear power plants), appreciation of the Bay, the simultaneous fragility and resilience of the ecosystem, active programs, or general assessment of conditions and needs.

Public involvement is substantial in several forums. The Chesapeake Bay Foundation (a member organization), Citizens' Program for Chesapeake Bay (an umbrella for groups), and the Maryland and Virginia Conservation Councils are highly active. The Chesapeake Bay Program includes one of the largest federal investments in public participation, supporting extensive forums, mini-grants, information exchange and a Citizens' Steering Committee to advise on the program (Wells et al., 1979; Davis, 1980).

Public opinion on water quality problems and other environmental issues is being assessed. Pollution was considered in 1971 to be of the greatest importance among Bay problems, especially from domestic wastes, industrial wastes, pesticides, and oil spills. Marylanders rated air and water pollution as serious environmental problems in 1979, with water pollution the more critical (Baltimore Environmental Center, 1980). A Sea Grant-sponsored telephone survey in 1979 found pollution to be the overriding concern of the citizens sampled with 69 percent ranking it first in relation to Chesapeake Bay (Florestano and Rathbun, 1980). Seventy-two percent disagreed with the statement that the Bay is in good shape. Eighty-four percent feared that an increase in waterfront industry would injure air and water quality. Vigorous punishment for dumping and pollution was supported. Among other results, the authors concluded that interest groups are generally in consonance with citizens' opinions, that few citizen users know about the active interest groups, and that many "interest groups" are in fact quite small sets of people organized to convey their views to agencies and to the media. An additional survey indicated that air and water pollution are of the greatest concern in Maryland, that industrial wastes are thought to be the major cause of water pollution, that nuclear facilities on the Bay were opposed by a majority, that citizens are uncertain whether improvements in environmental conditions have occurred in the past 10 years, but that they expect improvement in the next 10 (Rathbun and Linder, 1980).

RESEARCH NEEDS

There have been several catalogues of research needs for Chesapeake Bay, including those related to pollution (Beers et al., 1971; Chesapeake Research Consortium, 1977; Corps of Engineers, 1977; Ellis, 1973; Office of Research and Development/EPA, 1980). Eight areas of research and related activity merit emphasis:

1. *Improved comprehension of the components and fundamental processes of the total Chesapeake Bay system.* The physics, chemistry, and biology of the Bay have received much attention, but there are serious gaps that preclude the understanding necessary as a basis for adequate management. The routes, rates, sinks, and effects of materials entering the systems, including pollutants, are not sufficiently known. The food webs and flows of materials and energy in them are only grossly described. Knowledge of the requirements of the organisms of the Bay is so meager that only a small number of the 2,700 species larger than microorganisms can be carried through their full life history in laboratory culture.
2. *Development of technical methods for converting wasted resources from pollutants to useful materials.* Wasted heat and nutrients, chemical wastes, channel sediments, and the byproducts from processing of wood and foods are all large-scale pollutants of the Chesapeake Bay with such potentials.
3. *Research responsive to the needs recognized by management agencies for application in meeting short-term and long-term needs.* Competent and objective investigation of a large number of practical and managerial problems is needed. It must be protected from use as a cosmetic, mere defense of stated position, and the prostitution of the process of objective research.
4. *Improved understanding of the social and economic characteristics of the Bay region and improved capability to predict the effects of alternative practices.* Pollution control and other management efforts are anthropocentric, but there is a limited understanding of the social and economic needs, wishes, and trends of the region. Without it, the danger of misdirected management is great.
5. *Research to assist protection and enhancement of the desired conditions and products of the Bay, and to minimize undesired ones, by means consistent with the capacities and limitations of the ecosystem.* Improvement of fisheries by intelligent protection of their critical areas of spawning, nursery use, feeding, and migration is possible, but the knowledge base is very limited. Control of disease, of vegetation that interferes with use, of sea nettles, and of other components undesired by some users may become feasible, but we have at least learned that they must be approached in the context of balanced use and the total effects on a complex system. Knowledge is not yet sufficient to permit these achievements.
6. *Improved capability to predict the possible effects of proposed chemical introductions, physical alterations, biological modifications, and resultant environmental conditions in the Bay is essential for adequate management.* For example, we cannot state with sufficient precision to guide management agencies the effects, full costs, and full benefits of doubling or halving nutrient loads; releasing small quantities of toxicants or stimulants; cumulative construction of bulkheads, groins, small channels, and piers; large-scale extraction of commercially useful species; development of resistant bacteria; diminishing the freshwater flow from tributary rivers or altering the annual cycle of freshwater flow; or many other changes that have been, are, and will be proposed. Present knowledge can be very valuable in evaluation of such proposals, but it is not sufficient. Such research will involve basic studies, improved models of many kinds and, especially, improved means of experimentally testing the effects of change prior to risking a portion of the Bay. One of the most urgent needs for improving prediction of effects in Chesapeake Bay (and other

estuaries) lies in development of abilities to perform feasible laboratory bioassay tests and to predict realistically the impacts of the tested chemical or conditions on the principal organisms of the Bay and on the ecosystem. Present standard tests do not provide such prediction, except in very gross terms. Testing for acute and chronic effects, behavioral and physiological response measurement, extension of laboratory results to predict responses in the estuary—these and other difficult research areas must be explored far more fully.

7. *Development, testing, and permanent maintenance of excellent inventory and monitoring of the Bay.* Inventory of components and proper monitoring are needed to provide long-term descriptions, detect important changes, assist effective enforcement, and indicate vacancies where additional research is required. The vital signals of the system must be tracked for the total system and for the significant subdivisions—individual tributaries and inherently different segments of the open Bay.
8. *Development and use of comprehensive systems for the management and distribution of data and information.* Massive quantities of data are required and are now being produced. Better approaches are required, however, if they are to be adequate in scope, efficiently integrated, and readily available for retrieval, interpretation, and use. Similarly, technical and popular information must be better assembled, expressed, and distributed to users.

Assessment

The decade of the 1970s has been a period of exceptional change in pollution aspects of Chesapeake Bay, much of which favors improvement in protection of water quality. Notable federal legislation and a substantial body of recent state law has set new goals and established new mechanisms for regulation and enforcement. New and stronger management agencies have been created. Related research has been conducted at unpredicted levels of funding and sophistication. Analytic capabilities have improved until they sometimes threaten to overwhelm with information. Fresh assessments have been made of water quality in the estuary and in the watershed, and comprehensive projections have been essayed for the future 40 years. Maryland and Virginia have progressed from occasional reactive cooperation to establishment of positive and continuing interaction on Bay problems and needs at both executive and legislative levels. Public awareness and concern have been enhanced and support high environmental quality in the Bay.

However, not all changes in the 1970s demonstrate or promise improvements in environmental quality. Population growth continues at many sites, some of which are already heavily loading the estuary. Exotic toxic organic chemicals, to which the Bay biota is quite vulnerable, are continuously generated and sometimes released. The Kepone tragedy is a frightening example of failure of agencies and procedures established to protect the environment and a disturbing case of the extensive and enduring damage that is possible. Spills and near-spills (for example, the *Maria Costa*) continue, and shipping is expanding. A severe tropical storm demonstrated the capacity of natural events to overwhelm treatment facilities and management programs. Nuclear generating systems on the estuary and on the tributaries exemplify the very large scale of potential engineering changes and the massive uncertainties of the environmental effects of failure of a nuclear operation. Battles over power plant and refinery siting, placement of toxic dredged material and sewage sludge, and the management of sewage have not provided guarantees that environmental quality will be assured in these and other matters.

A principal protection device, the National Pollution Discharge Elimination System, appears to be grossly inadequate in describing or controlling pollution from industrial and domestic systems. Nutrient concentrations of threatening levels have been shown to be progressing down the tributaries and the main stem of the Bay. Conversion of wastes to useful resources has not been adequately accomplished.

Several hangovers from the past exist. The long-term accumulations of known toxicants and probably of unknown ones, especially in Baltimore Harbor and the Elizabeth River, exist as complex reservoirs of pollutants, readily available for release and access to the biota, as does Kepone in the James. In old domestic and industrial systems, some ancient pipes, valves, outfalls, and practices are difficult to correct and expensive to replace. At many sites waste treatment capacity and practice fail to stay ahead of loading. Substantial reduction has occurred in the 1970s in numbers of important Bay plants and animals, and the role of water quality has not been determined.

Uses are not yet rationally zoned in accordance with the primary characteristics and capacities of the Bay system, and means for controlling the distribution and density of human populations are limited and local. The enormous role of land usage in affecting water quality is rather dimly seen and does not yet affect many decisions to permit or refuse various land-based activities.

There is no continuing source of funding for long-term research and monitoring designed to comprehend the Bay and assist in its management.

On balance, the potentials for achieving and sustaining high water quality in the tidal system of the Chesapeake Bay have been greatly enhanced and they may possibly be realized in the next decade or two. However, very serious problems remain, and the obstacles to such achievement are formidable.

The vast values of the Chesapeake for many uses merit conservative guarantees, within the limits of human capability, that the environmental quality, inherent processes, and biological health of the Chesapeake Bay will indeed be assured for the indefinite future. Almost every important and desired use depends on that achievement. Recent progress is impressive. Further progress is imperative.

REFERENCES

American Water Resources Association, National Capitol Section. 1976. Symposium—the future of Chesapeake Bay. Am. Water Res. Assoc., Nat. Cap. Sect.

Anonymous. 1980. Sludge: that slimy goo is very hard to get rid of. Baltimore Evening Sun. October 3, 1980.

Associated Press. 1980. Recreational fishermen can keep Kepone-filled fish. Baltimore Evening Sun. September 24, 1980.

Austin, H. 1979. Major environmental impact affecting Virginia fisheries. Part 1. Conference on Virginia Fisheries and the Environment, B. Theberge, ed. Marshall-Wythe School of Law, College of William and Mary. pp. 21-27.

Baltimore Environmental Center. 1980. Environmentalism enters the 80's: public opinion in an era of policy trade-offs. Baltimore Environmental Center.

Beers, R.F., F. Bender, L.M. Brush, Jr., R.J. Byrne, L.E. Cronin, R.L. Green, W.J. Hargis, Jr., E.B. Joseph, L.C. Kohlenstein, R.W. Krauss, R.H. Roy, A.B. Schulz, and F.T. Sparrow. 1971. The Chesapeake Bay—a research program to assist in better management of complex environmental systems. The Johns Hopkins University, University of Maryland, and Virginia Institute of Marine Science.

Bellanca, M.A. In press. Management implications of nutrient standards for estuaries. Enrichment of estuaries, B.J. Neilson and L.E. Cronin, eds. 1980.

Bender, M.E., R.J. Huggett, and H.D. Slone. 1972. Heavy metals—an inventory of existing conditions. J. Wash. Acad. Sci. 62(2):144-156.

Bergoffen, W. 1971. Conference report. Citizens' program for the Chesapeake Bay. Natural Resources Institute, University of Maryland. Ref. 71–77. Mimeo.

Bertine, K., J.K. Cochran, L.E. Cronin, W.P. Davis, C.S. Martens, L.R. Pomeroy, J. Schubel, J.L. Taft, J. Teal, and R. Wilson. 1980. Estuaries. Assimilative capacity of U.S. coastal waters for pollutants. Proceedings of a workshop, E. Goldberg, ed. Working Paper No. 1: Federal Plan for Ocean Pollution Research, Development, and Monitoring. FY 1981–1985. Environmental Research Labs, NOAA, U.S. Department of Commerce. pp. 59–97.

Block, R.M., and G.R. Helz. 1977. Proceedings of the chlorination workshop. Sixteen papers on chlorine use, analysis of residuals, organochlorides, bioassay, and physiological responses related to chlorine. Chesapeake Science, 18(1):93–160.

Brush, L. 1974. Inventory of sewage treatment plants for Chesapeake Bay. Chesapeake Research Consortium, Publication No. 28.

Chambers, J. 1979. Major environmental impacts affecting Virginia fisheries. Part 2. Conference on Virginia Fisheries and the Environment, B. Theberge, ed. Marshall-Wythe School of Law, College of William and Mary. pp. 27–31.

Champ, M.A. Nutrient loading in the nation's estuaries. Estuarine pollution control and assessment. Proceedings of a conference, Office of Water Planning and Standards, U.S. Environmental Protection Agency, 1:237-256.

Chesapeake Bay Foundation. 1977. The Bay on borrowed time: transportation and handling of oil and other hazardous materials on Chesapeake Bay waters. Chesapeake Bay Foundation.

Chesapeake Research Consortium. 1976. The effects of Tropical Storm Agnes on the Chesapeake Bay System. Summary and 47 technical reports on hydrological effects, geological effects, water quality effects, biological effects, economic impacts, and public health impacts. The Johns Hopkins University Press, Baltimore.

Chesapeake Research Consortium. 1977. Proceedings of the Bi-State Conference on the Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61.

Chesapeake Research Consortium. 1978. Chesapeake Bay directory. Chesapeake Research Consortium, Publication No. 63.

Citizens' Program for the Chesapeake Bay, Inc. 1978. Forum on herbicides in the Chesapeake Bay. Mimeo.

Coastal Ecosystem Project, Office of Biological Services, U.S. Fish and Wildlife Services. In press. National Symposium on Freshwater Inflow into Estuaries, September 9–11, 1980.

Colwell, R.R. 1977. Bacteria and viruses—indicators of unnatural environmental changes occurring in the nation's estuaries. Estuarine pollution control and assessment. Proceedings of a conference, Office of Water Planning and Standards, Vol. II. U.S. Environmental Protection Agency. pp. 507–518.

Colwell, R.R., and J. Kapper. 1978. Distribution, survival and significance of pathogenic bacteria and viruse in estuaries. Estuarine interactions, M. Wiley, ed. Academic Press, New York. pp. 443-457.

Congress of the United States. 1980. Chesapeake Bay Research Coordination Act of 1980. 96th Congress, 2d Session.

Corps of Engineers, Baltimore District. 1974. Chesapeake Bay existing conditions report.

Corps of Engineers, Baltimore District. 1977. Chesapeake Bay future conditions report.

Cronin, L.E. 1967. The condition of the Chesapeake Bay. Thirty-second Wildlife and Natural Resources Conference, Wildlife Management Institute. pp. 137–150.

Cronn, L.E. Ecology. 1976. Symposium on the future of Chesapeake Bay. Am. Water Res. Assoc., Nat. Cap. Sect. pp. 93-111.

Cronin, L.E. 1976. The effects of increased oil tanker traffic on the Chesapeake Bay. Citizens' Program for Chesapeake Bay. Mimeo.

Cronin, L.E. 1978. A biologist views Chesapeake Bay. Proceedings of the 68th Convention, International Association of Fish and Wildlife Agencies. pp. 7-12.

Cronin, L.E. 1979. Testimony for the Chesapeake Research Consortium on coordination of research related to Chesapeake Bay. Hearing before the Subcommittee on Governmental Efficiency and the District of Columbia, Committee on Governmental Affairs, U.S. Senate, 96th Congress, 1st Session.

Cronin, L.E., M.G. Gross, M.P. Lynch, and J.K. Sullivan. 1977. The condition of the Chesapeake Bay—a consensus. Proceedings of the Bi-State Conference on Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:37–57.

Cronin, L.E., M.P. Lynch, and M.J. Karweit. 1979. Data importance in relation to Chesapeake Bay pollution. Proceedings of the 6th International Codata Congress, B. Dreyfus, ed. Pergamon Press, Oxford. pp. 21–30.

Cronin, L.E., D.W. Pritchard, T.S.Y. Koo, and V. Lotrich. 1977. Effects of enlargement of the Chesapeake and Delaware Canal. In: Estuarine processes, Vol. II, M. Wiley, ed. Academic Press, NY. pp. 18–22.

Cronin, L.E., D.W. Pritchard, J.R. Schubel, and J.A. Sherk, eds. 1974. Metals in Baltimore Harbor and upper Chesapeake Bay and their accumulation by oysters. Chesapeake Bay Institute, The Johns Hopkins University; and Chesapeake Biological Laboratory, University of Maryland.

Cronin, W.B. 1971. Volumetric, areal and tidal statistics of the Chesapeake Bay estuary and its tributaries. Chesapeake Bay Institute, The Johns Hopkins University, Special Report 20.

Davies, T.T. 1980. EPA Chesapeake Bay program. Personal communication.

Davis, W.P., and D.P. Middaugh. 1977. Impact of chlorination on marine ecosystems. Estuarine pollution control and assessment. Proceedings of a conference, Office of Water Planning and Standards, Vol. II. U.S. Environmental Protection Agency. pp. 415–423.

Douglas, J.E., Jr. 1979. Summary report of the select inter-agency task force on chlorine. Va. Mar. Res. Comm.

Ellis, R.H. 1973. A method for applying research to environmental planning and management: a case study of issues important to the Chesapeake Bay. The Center for the Environment and Man, Inc.

Farrington, J.W. 1977. Oil pollution in the coastal environment. Estuarine pollution control and assessment. Proceedings of a conference, Office of Water Planning and Standards, Vol. II. U.S. Environmental Protection Agency. pp. 385-401.

Federal Water Pollution Control Administration. 1969. The national estuarine pollution study. Chesapeake Bay. 3:V-265–V-285.

Fish and Wildlife Service. 1970. National Estuary Study. 7 vol. Vol. 3, Chesapeake Bay. pp. 65-112.

Fisher, A.C., Jr. 1980. My Chesapeake—queen of bays. National Geographic 158(4):428-467.

Flemer, D.A., and D.R. Heinle. 1974. Effects of wastewater on estuarine ecosystems. Chesapeake Research Consortium. Publication No. 33.

Florestano, P.S., and P.A. Rathbun. 1980. The relationship between resource users and interest group representation on the Chesapeake: a summary. University of Maryland Sea Grant Project Report. Mimeo.

Friedlander, S. 1979. Jurisdictional breakdown of the Chesapeake Bay by federal agencies. Report to the Chesapeake Bay Legislative Advisory Commission. Md. Dept. Leg. Ref. Mimeo.

Garreis, M.J. 1980. Acres of Maryland shellfish waters closed on basis of bacterial survey, 1970– 1980. Maryland Department of Health and Mental Hygiene. Mimeo.

Gartlan, J.V., Jr., L.E. Cronin, et al. 1980. Report of the Chesapeake Bay Legislative Advisory Commission to the General Assemblies of the Commonwealth of Virginia and the State of Maryland.

GCA/Technology Division. 1979. Inventory and toxicity prioritization of industrial facilities discharging into Chesapeake Bay. Contract Report to Inventory/Prioritization Workshop, U.S. Environmental Protection Agency.

Green, K.A. 1978. A conceptual ecological model for Chesapeake Bay. Coastal Ecosystems Project, Office of Biological Services, U.S. Fish and Wildlife Services. FWS/OBS 78/69.

Hedgepeth, J. 1972. Symposium summary—the fate of the Chesapeake Bay. J. Wash. Acad. Sci. 62(2):218–223.

Heinle, D.R., J.L. Taft, C.F. D'Elia, J.S. Wilson, M. Cole-Jones, A.B. Vivian, and L.E. Cronin. In press. Historical review of water quality and climatic data from Chesapeake Bay with emphasis on enrichment. EPA Chesapeake Bay Program Report. Chesapeake Research Consortium, Publication No. 84.

Hess, R.E., R.E. Bowles, V. Keith, N.G. Kelly, P.J. O'Connor, J. Porricelli, H. Schwartzman, E.C. Weber, and R. Wilcox. 1977. Workshop report on prevention and control of spills. Proceedings of the Bi-State Conference on Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:141–163.

Hoffman, D. 1979. Who's killing the Chesapeake Bay? The Washington Post Magazine. April 29, 1979.

Horn, D.A., N. Doelling, D.A. Levey, B. Davies, R.W. King, L. Koppelman, F. Monastero, B. Neilson, and J.R. Schubel. 1980. Report of the North and Mid-Atlantic Regional Conference on Ocean Pollution Research, Development and Monitoring. MIT Sea Grant College Program.

Horton, T. 1980. Water treatment with chlorine blamed for decline in Bay life. Baltimore Sun, July 14, 1980.

Huggett, R.J., and M.E. Bender. In press. Pesticides in the James River Estuary. Environmental Science and Technology.

Huggett, R., R.M. Block, O. Bricker, T. Felvey, and G.R. Helz. 1977. Workshop Report on Toxic Substances. Proceedings of the Bi-State Conference on Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:121–127.

Huggett, R.J., M.M. Nichols, and M.E. Bender. 1980. Kepone contamination of the James River Estuary. R.A. Baker, ed. Contaminants and sediments, Vol. I. pp. 33-52. American Chemical Society, Washington, DC.

Huggett, R.J., O.P. Bricker, G.R. Helz, and S.E. Sommer. 1974. A report on the concentration, distribution and impact of certain trace metals from sewage treatment plants on the Chesapeake Bay. Chesapeake Research Consortium, Publication No. 31.

Hydroscience, Inc. 1975. The Chesapeake Bay waste load allocation study. Report prepared for the Water Resources Administration, Maryland Department of Natural Resources.

Janssen, W.A., and C.D. Meyers. 1968. Fish: serologic evidence of infection with human pathogens. Science 159:547–548.

Jaworski, N.A. In press. Sources of nutrients and the scale of eutrophication problems in estuaries. In: Enrichment of estuaries, B. Neilson and L.E. Cronin, eds. Humana Press, Clifton, NJ.

Jensen, E.T. 1976. Pollution. Symposium on the Future of Chesapeake Bay. American Water Resources Association, Nat. Cap. Sect. pp. 66–92.

Kanigel, R. 1979. How fragile is the Bay? Johns Hopkins Magazine 30(4):28-36.

Kelly, N.G. 1976. The estimated impact of a hypothetical oil spill off Smith Point, Virginia. Chesapeake Bay Foundation.

Kohlenstein, L.C. 1972. Systems for storage, retrieval and analysis of data. Chesapeake Science 3(suppl):S157–S168.

Kuo, A.Y., A. Rosenbaum, J.P. Jacobson, and C.S. Fang. 1975. Analysis and projection of water quality. In: The Chesapeake Bay: a study of present and future water quality and its ecological effects, Vol. I. Report to the National Commission on Water Quality. Virginia Institute of Marine Science.

Laniak, G.F. A nutrient balance of the Chesapeake Bay: with application to monitoring of nutrients. Chesapeake Bay Program, Reg. 3, U.S. Environmental Protection Agency, Nutrient Work Paper No. 1.

LeBlanc, N.E., M.H. Roberts, Jr., and D.R. Wheeler. 1978. Disinfection efficiency and relative toxicity of chlorine and bromine chloride: a pilot plant study in an estuarine environment. Special Report Applied Marine Science, No. 206, Virginia Institute of Marine Science.

Lippson, A.J., ed. 1973. The Chesapeake Bay in Maryland: an atlas of natural resources. The Johns Hopkins Press, Baltimore.

Lippson, A.J., M.S. Haire, A.F. Holland, F. Jacobs, J. Jensen, R.L. Moran-Johnson, T.T. Polgar, and W.A. Richkus, 1980. Environmental atlas of the Potomac Estuary. Environmental Center, Martin Marietta Corporation.

Lippson, R.L., and A.J. Lippson. 1979. The condition of Chesapeake Bay—an assessment of its present state and its future. Presented to Marine Environmental Quality Committee, Int. Counc. for Expl. Sea, Warsaw, Poland. Mimeo.

Lunsford, C.A., C.L. Walton, and J.W. Shell. 1980. Summary of Kepone study results, 1976-1978. Va. State Water Control Bd. Basic Data Bull. No. 46.

Lynch, M.P., W.F. Conley, T.S.Y. Koo, G. Krantz, R. Lippson, J. Merriner, J.F. Percival, and W.R. Prier. 1977. Workshop Report on Fisheries and Wildlife. Proceedings of the Bi-State Conference on the Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:189-209.

McErlean, A.J., C. Kerby, and M.L. Wass, eds. 1972. Biota of the Chesapeake Bay. Includes overview articles on the biota; summaries for 27 large taxa; status reports on effects of sediment, nutrients, metals and pesticides; sample inventories for 8 species or groups; and other material. Chesapeake Science 13 (suppl.).

McGarry, R.S. 1976. Channelization and shipping. Symposium on the Future of Chesapeake Bay. American Water Resources Association. Mimeo. pp. 20–40.

McKay, J.H., Jr. 1976. The hydraulic model of Chesapeake Bay. In: Estuarine processes, Vol. II, M. Wiley, ed. Academic Press, NY. pp. 404–415.

McKewen, T.D. 1972. Human wastes and the Chesapeake Bay. J. Wash. Acad. Sci. 62(2):157-161.

Mihursky, J.A., and J.B. Pearce, eds. 1969. Proceedings of the 2nd Thermal Workshop of the U.S. International Biological Program. Chesapeake Science. 10(3–4).

National Aeronautics and Space Administration. 1972. Remote sensing of the Chesapeake Bay. Report of a conference. Includes 23 talks plus reports from group discussions. NASA SP-294.

National Aeronautics and Space Administration. 1978. Application of remote sensing to the Chesapeake Bay region. Vol. 2, proceedings, W.T. Chen, G.W. Freas, G.D. Hickman, D.A. Pemberton, T.D. Wilkerson, I. Adler, V.J. Laurie, eds. 26 papers, 9 group reports, 3 resource contributions.

Nichols, M.M., N.H. Cutshall, and R.C. Trotman. 1979. Tracing Kepone contamination in James Estuary sediments. Workshop on Sediment and Pollution Interchange in Shallow Seas, Int. Counc. Expl. Sea.

Office of Research and Development, U.S. Environmental Protection Agency. 1980. Chesapeake Bay. Research Summary. USEPA-600/8-80-019.

Office of Water Planning and Standards. 1977. Evaluation of the problem posed by in-place pollutants in Baltimore Harbor and recommendation of corrective action. USEPA 440/5-77-015A&B.

Perkinson, W.J., B. Burton, and D. Mills. 1973. The Chesapeake at bay. Illustrated report on water pollution. The Baltimore Evening Sun.

Pheiffer, T.H., D.K. Donnelly, and D.A. Possehl. 1972. Water quality conditions in the Chesapeake Bay system. Annapolis Field Office, Reg. III, EPA, Tech. Rept. 55.

Prentiss, L.W., Jr. 1972. The Corps of Engineers Chesapeake Bay Study. J. Wash. Acad. Sci. 62(2):190-195.

Pritchard, D.W. 1968. Chemical and physical oceanography of the Bay. Proc. Gov. Conf. on Ches. Bay:II-49—I-74.

Pritchard, D.W. 1971. A brief comment on the most serious water quality problems of the Chesapeake Bay and tributary tidal waterways. The estuarine environment—estuaries and estuarine sedimentation. AGI Short Course Lecture Notes. Mimeo. pp. XV 9-11.

Pritchard, W.E. 1980. Marine Science Research Center, State University of New York, Stony Brook. Personal communication.

Rathbun, P., and G. Linder. 1980. Attitudes toward the environment: a survey of Maryland's citizens. Prepared for the Maryland Coastal Zone Management Program. Maryland Opinion Survey, University of Maryland.

Roberts, M.H., Jr., D.F. Boesch, and M.J. Bender. 1975. Analysis and projection of ecological conditions. In: The Chesapeake Bay: a study of present and future water quality and its ecological effects, Vol. II. Report to the National Committee on Water Quality. Virginia Institute of Marine Science.

Roberts, M.H., Jr., R.J. Diaz, M.J. Bender, and R.J. Huggett. 1975. Acute toxicity of chlorine to selected estuarine species. J. Fish. Res. Bd. Can. 32(12):2525-2528.

Roberts, M.H., Jr., C.E. Laird, and J.E. Illowsky. 1979. Effects of chlorinated seawater on decaped crustacean and *Mulinia* larvae. EPA-600/3-79-031. Env. Res. Lab., U.S. Environmental Protection Agency, Gulf Breeze, FL.

Robinson, A.E. 1980. Chesapeake Bay study. Corps of Engineers, Baltimore District, Personal communication.

Rose, C.D. 1974. Petroleum in the estuary. NRI Special Report No. 5. Contr. 584, Natural Resources Institute, University of Maryland.

Schubel, J.R. 1972. The physical and chemical conditions of the Chesapeake Bay. J. Wash. Acad. Sci. 62 (2):56–72.

Schubel, J.R., and H.H. Carter. 1976. Suspended sediment budget for Chesapeake Bay. In: Estuarine processes, Vol. II, M. Wiley, ed. Academic Press, NY. pp. 48–62.

Schubel, J.R., and A.D. Williams. 1976. Dredging and its impacts on upper Chesapeake Bay: some observations. Time-stressed coastal environments: assessment and future action. Proceedings of the 2nd Annual Conference on Coastal Soc. pp. 70–115.

Schubel, J.R., and W.M. Wise, eds. 1979. Questions about dredging and dredged material disposal in the Chesapeake Bay. Marine Science Center, S.U.N.Y. Special Report 20.

Shea, G.B., G.B. Mackiernan, L.C. Athanas, and D.F. Bleil. 1980. Chesapeake Bay low flow study:biota assessment. Phase I: final report. Western eco-systems technology. 2 vol.

Snedaker, S., D. deSylva, and D. Cottrell. 1977. A review of the role of freshwater in estuarine ecosystems. Final report to the Southwest Florida Water Management District. Rosen. School of Marine and Atmospheric Science, University of Miami.

State of Maryland. 1968. Proceedings of the Governor's Conference on Chesapeake Bay.

State of Maryland and Commonwealth of Virginia. 1980. Chesapeake Bay striped bass research program. Introduction. Mimeo.

Stevenson, J.C., and N.M. Confer. 1978. Summary of available information on Chesapeake Bay submerged vegetation. Coastal Ecosystems Team, Office of Biological Services, U.S. Fish and Wildlife Service. FWS/OBS - 78/66.

Sullivan, J.K., E.R. Collins, D.L. Correll, P. Fisher, J.C. Kirsch, T.H. Pheiffer, and R.F. Schoenhofer. 1977. Workshop report on non-point pollution. Proceedings of the Bi-State Conference on the Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:223-240.

Tsai, C. 1975. Effects of sewage treatment plant effluents on fish: a review of literature. Chesapeake Research Consortium, Publication No. 36.

Tsai, C., J. Welch, K. Chang, J. Schaeffer, and L.E. Cronin. 1979. Bioassay of Baltimore Harbor sediments. Estuaries 2(3):141–153.

Ulanowicz, R.E. 1976. Modeling the Chesapeake Bay and its tributaries: a synopsis. Ches. Sci. 17(2):114–122.

Villa, O., F. Hamons, C.G. Hill, M. Moriarty, A.E. Robinson, L.W. Willett, F.T. Wootton, and J.M. Zeigler. 1977. Workshop report on maritime development. Proceedings of a Bi-State Conference on Chesapeake Bay. Chesapeake Research Consortium, Publication No. 61:65–96.

Virginia Marine Resources Commission. 1979. Proceedings of a seminar held by the Select Inter-Agency Task Force on Chlorine. Va. Inst. Mar. Sci. Mimeo.

Wallace, McHarg, Roberts and Todd, Inc. 1972. Maryland Chesapeake Bay study.

Walsh, G.E. 1972. Insecticides, herbicides and polychlorinated biphenyls in estuaries. J. Wash. Acad. Sci. 62(2):122–139.

Washington Academy of Sciences. 1972. Symposium—science and the environment (II): the fate of the Chesapeake Bay. J. Wash. Acad. Sci. 62(2) :51-223.

Wells, H.W., W.C. Allen, and H.E. Rector. 1979. The Chesapeake Bay Program: project summaries. Chesapeake Bay Program, Office of Research and Development and Region III, U.S. Environmental Protection Agency.

Wiley, C.W. 1980. Condemned shellfish areas, 1970-1980. Bureau of Shellfish Sanitation, Virginia State Department of Health. Personal communication.

Williamson, F.S.L. 1972. Biology and the Chesapeake Bay. J. Wash. Acad. Sci. 62(2):88–102.

Wilson, H.O. 1977. Letter of June 22, 1977, to W.D. Johnston, III.

Withers, G.K. 1979. Testimony to Subcommittee on Governmental Efficiency and the District of Columbia, Committee on Governmental Affairs, U.S. Senate, 96th Congress.

POLLUTION IN THE NEW YORK BIGHT: A CASE HISTORY

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DESCRIPTION OF THE BIGHT

Physical Features

Geographically, the New York Bight is a relatively sharp angle in the northeastern U.S. continental shelf south of Long Island, New York, and east of New Jersey. The shelf is about 180 km (112 mi) wide and has an areal extent of about 39,000 km^2 (15,000 n mi^2). Depths over much of the Bight are between 30 to 60 m (98 to 197 ft), and the shelf break is defined at a depth of 200 m (656 ft). The broad, gently sloping shelf of the Bight is bisected by the Hudson Shelf Valley.

The morphology of the Bight floor and the distribution of its surficial sediments are controlled by sea-level fluctuations from continental glaciation over the past several million years. At the time of the last major ice advance, the North American ice sheet extended from Canada to Long Island and northern New Jersey. Sea level was lowered to about 145 m (480 ft) below the present level about 15,000 years ago; hence, the continental shelves became dry land. Since then, the ice has been melting and the shoreline has retreated over the shelf to its present position. Many features on the shelf today are the result of this fall and rise of sea level (Freeland and Swift, 1978).

Extensive sampling of surficial sediments has provided rather detailed knowledge of grain size distribution. The dominant material on the shelf floor is sand, 0 to 10 m (0 to 33 ft) thick, resting on Holocene clays. Unconsolidated fine sediments are regularly resuspended and ultimately carried back into the estuaries or off the shelf edge. Some persistent areas of muds or fine sediments occur in low areas, in the Christiaensen Basin, Hudson Shelf Valley, and in smaller "mud patches" near Long Island. The muds accumulate in relatively quiet, deep areas because these regions have less intense wave and current energy. Nearshore mud patches develop because high concentrations of suspended sediments are available for deposition during calm periods (Freeland and Swift, 1978).

Fine sediments enter the Bight by discharge from rivers and tidal inlets, and by dumping, particularly of dredged materials that have accumulated in New York Harbor. Domestic and industrial wastes from land also contribute to elevated concentrations of organic carbon and toxic metals (Swanson, 1977) and a wide variety of synthetic organic compounds (O'Connor, J.M., et al., 1981) in the sediments.

Water movements in the Bight are highly variable. Over the middle and outer shelf, waters generally move to the south-southwest, parallel to the bathymetric contours. The average flow is about 5 cm/s (0.1 knots) near the surface, reduced to about 1 cm/s (0.02 knots) near the bottom. On the outer shelf, storm-induced winter currents of 3 to 10 days are common (Mayer, et al., 1979). Water temperature follows the well-known seasonal cycle of heating and stratification in summer and vertical homogeneity in winter.

While boundaries between the inner and outer Bight are poorly defined and constantly changing, the inner Bight does have two important features that tend to limit its capacity to flush contaminants to the open ocean:

1. The two-layer flow near the mouth of New York Harbor is dominated by flow from the Hudson-Raritan estuary. The less dense surface layer flows seaward, generally parallel to the New Jersey coast. The lower, denser water of the Bight flows into the estuary. Fine sediments that are rich in pollutants and organic carbon tend to sink, be entrained in the bottom waters, and undergo the partially closed cyclical transport common in estuaries.
2. East of the region of strong river influence, a clockwise circulation gyre is evident for part of the time. Its western edge tends to be located over the head of the Hudson Shelf Valley. While water flow can be up or down valley, up-valley flow has been measured for extended periods of time (Beardsley, et al., 1976). These impermanent up-valley flows and the clockwise gyre tend to reduce the flushing of contaminants from the inner Bight or apex.

Historical Changes

The broad historical trends in human influences on the Bight are functions of striking increases in population density and energy usage. The human population of the counties bordering the Bight increased exponentially from about 10,000 in 1675 to over 16 million in 1970. Population stabilized in the period from 1970 to 1980 for the first time since American Indians were driven from the region by Europeans in the 1600s (O'Connor, J.S., 1981).

From the 1850s to 1940, energy usage very gradually increased in the coastal fringe of the Bight. Even in the early 1800s, however, the New York metropolitan region was established as the rapidly expanding focus for development in the United States. By 1810, New York City's population exceeded that of Boston, and the city became the largest in the nation. Prosperous European nations invested heavily with capital, goods, and technology in North America, and New York was a principal beneficiary of this investment through trade, services, and manufacturing (Squires, 1981). The Port of New York reached its peak in 1871 as an import/export center, handling 71 percent of the nation's foreign trade, but as this share declined, New York's commercial, financial, and industrial growth continued (Boddewyn, 1981).

Beginning in the 1940s, energy consumption soared. This spurt of energy consumption coincided with continuing growth in population and (with some exceptions) industry and services through the 1960s (O'Connor, J.S., 1981).

Until the mid-1700s, garbage and other wastes were dumped into open gutters, rivers, or onto land to feed pigs and chickens. These practices continued into the 1850s in some areas while sewers and cesspools were being constructed. By 1806 it was clear to the New York City Board of Health that more extensive sewers needed to be constructed throughout the populated areas and that the city aquifers could no longer provide enough potable water (Loop, 1964). By 1842 the original Croton aqueduct carried water to the city from 60 km (37 mi) to the north. This aqueduct was replaced and extended to 150 km (90 mi) from the city in 1915, and another aqueduct brought freshwater from the Delaware basin by 1944. The system's 11 reservoirs provide apparently safe yields of at least 565 m^2/s (1,290 mgd) even during the 1961–1967 drought. Per capita consumption of water has increased from 380 l/capita/day (100 gcd) in 1900 to 660 lcd (175 gcd) in 1977 (Gunnerson, 1981). This rapid increase in usage of water for domestic and industrial purposes has led to increasingly difficult wastewater disposal problems.

By 1972 separate and combined sewers were serving most of the New York/New Jersey metropolitan area and parts of the New Jersey coast. These sewers contributed 114 m^3/s (2.6 mgd) of municipal discharges to the Hudson-Raritan estuary and New York Bight. Industrial discharges were 27 m^3/s (0.61 mgd) of which 47 percent went through municipal systems. Total wastewater flows (domestic, industrial, and urban runoff) to the Bight averaged 173 m^3/s (3.95 mgd). These wastewaters represented about 22 percent of the total freshwater flow (790 m^3/s or 18 mgd) to the Bight (Mueller, et al., 1976).

So-called solid wastes, some of which are primarily water, have historically been dumped on (then) lower valued lands to create new land through shoreline extension, and at harbor and ocean dump sites at increasing distances from population centers. Prior to 1970 about 1.4 billion m^3 (1.9 billion yd^3) of total waste solids were dumped in New York waters. This amount exceeded the suspended sediment discharge of all the Atlantic coast rivers (Gross, 1976).

Street sweepings, garbage, and refuse were dumped first in New York Harbor and from 1900 to 1934 in the inner Bight. However, in response to the garbage washing up on the beaches, a Supreme Court decision forbade further ocean discharge of garbage. Since 1934 these floatable wastes have been incinerated or landfilled (Gross, 1976).

Sediments have been dredged from ship channels of New York Harbor since at least the early 1800s. Sewage sludges and acid wastes dumped in the Bight are primarily liquid. They contain about 5 percent and <1 percent solids, respectively. The sludges from sewage treatment plants have been dumped at sea since 1924 (Gross, 1976) and acid wastes since 1948 (MESA, 1975). Relatively nontoxic solid wastes from construction and demolition have also been dumped at a specific ocean dump site since the early 1800s (Gross, 1976).

The volumes of domestic and industrial wastes of the Bight region increased substantially as the 1970s approached. Two changing characteristics of this waste stream posed increasingly difficult problems for what was to become residuals management. First, the wastewaters became increasingly toxic as oils, toxic metals, and then synthetic organic compounds were discarded. Secondly, the ecologically hazardous wastes had become diluted in volumes of water that could be treated, even superficially, only at great cost.

Ecosystems Structure and Function

The structure and ecosystem productivity of the Bight are generally comparable to those of other temperate coastal environments. Averaged over the entire Bight, the biomass of primary producers and herbivores is about 2 g C/m^2 each, carnivore biomass is about 6 g C/m^2, and the biomass of decomposers averages about 4 g C/m^2 (O'Connor, J.S., 1981). The annual primary productivity (200 to 300 g $C/m^2/yr$) organic content of shelf sediments (0.5 to 1%, dry wt.) and fish yields (10 tons/km^2/yr) is comparable to that of other mid-latitude continental shelves (Walsh, 1980). However, the inner Bight has been perturbed by human activity resulting in elevated primary productivity and sediment organic concentrations, diseased fish and shellfish, prohibitions against harvesting of filter feeding shellfish because of pathogen contamination, and other biotic effects (O'Connor, J.S., 1976; 1981).

The planktonic organisms of the taxonomic groups characteristic of inshore waters are typically more abundant and productive than those of offshore waters, particularly in the apex, which is enriched by organic carbon and nutrient wastes from several sources. Phytoplankton productivity of the apex, for instance, averages about three times higher than that of the outer Bight (Malone et al., 1979).

The predominantly sandy and muddy sand sediments of the Bight are particularly hospitable environments for four groups of larger benthic invertebrates: bivalve molluscs, annelid worms, the echinoids (sea urchins and sand dollars), and crustacean shellfish. Both the total numbers of benthic invertebrates and their total weight decrease markedly from nearshore to the edge of the continental shelf. Benthic biomass depends as well upon sediment type, increasing by a factor of almost 20 from sand-gravel (94 g/m^2) to silty sand (1,800 g/m^2) (Wigley and Theroux, 1976).

As a consequence of long-term carbon and toxicant loadings to the inner Bight, an area of sediments greater than 240 km^2 (93 m^2) in the apex is enriched with carbon, toxic metals, petroleum hydrocarbons, and synthetic organic compounds. These same sediments typically contain depauperate benthic communities, with very high standing crops of a very few species. Organic enrichment of fine-grained sediments seems to be the major factor altering the preexisting competitive balance among many species whose feeding strategies differ. The high standing stocks and reduced species diversity observed in parts of the apex may be caused by this mechanism. Additional stresses that may contribute to these effects include the production of toxic sulfide ions and resuspension of other toxicants that tend to exclude predators and reduce cropping. Until recently, observations of benthic community alterations had been restricted primarily to the benthic macrofauna. However, recent studies have shown parallel disturbances in meiofaunal species assemblages of affected sediments (Tietjen, 1980).

Fish populations of the Bight are dominated by migratory species. Temperature is a strong stimulus for the migration of most coastal fishes, and temperature changes also stimulate spawning. The abundance of individual fish species within the Bight has commonly fluctuated by a factor of four in the past 25 years. While the major cause of year to year fluctuations is climatic variability, sport and commercial fishing caused serious declines in nearly all commercially important species from 1960 to the mid-1970s. From 1967 to 1974 the total biomass of finfish caught by bottom trawls declined by more than 50 percent in the region from Cape Cod to Cape Hatteras. Sport fishing has also increased rapidly. Recreational catches are estimated to be about as large or larger than domestic commercial catches for striped bass, bluefish, weakfish, summer flounder, winter flounder, black sea bass, cod, and mackerel. Since 1974 more restrictions have been imposed to reduce commercial fishing levels, and fish stocks as a whole have increased (Grosslein and Azarovitz, 1981). The most significant finfishes in the Bight for commercial and sport fishing demands include cod, summer flounder, bluefish, striped bass, Atlantic mackerel, winter flounder, black sea bass, and weakfish.

Shellfish are also important commercial and recreational resources. Based upon sampling in 1976, the estimated biomass of ocean quahogs was 2,450,000 t of meats; surf clam meats were estimated at 875,000 t. Sea scallops are another major offshore resource for which biomass estimates are not available.

The food webs built up from plant material and detritus lead to continual replenishment of harvestable fish and shellfish resources of the Bight. However, within the Hudson-Raritan Estuary several fish and shellfish species no longer grow and reproduce adequately to sustain exploitation. Even those species maintaining harvestable densities within the estuary contain PCB concentrations that approach or exceed FDA limits and cannot be harvested commercially. Filter feeding shellfish within the estuary and the inner Bight contain concentrations of coliform bacteria that prohibit commercial exploitation.

The massive discharges of particulate material from the New York metropolitan region both limit the primary productivity within the estuary and stimulate the detrivores that feed upon organic particles. However, the limited information available indicates that the rates of carbon degradation within the estuary have not increased commensurate with increases in historical organic carbon loadings. The excess organic carbon has both accumulated in the estuarine sediments and been transported to the Bight. The aesthetic and recreational value has been compromised by oil sheens, floatable wastes, and odors that are widespread within the estuary. To a lesser extent, oil sheens and floatable wastes are frequent features of the Bight as well.

The most important economic function of the Bight and estuary is as a vehicle for coastal and foreign transport. The total foreign and domestic waterborne commerce to and from the Port of New York was about 175 × 10^6 t/yr during the 1970s.

RECENT MODIFICATIONS OF THE ECOSYSTEM

Environmental Issues — 10 Year Changes

Few striking trends over the past 10 years are evident in the structure or function of the ecosystem. This lack of evidence may well be due partially to the combination of limited measurements and the large variability in biotic responses to natural environmental fluctuations. Only a few attributes of the Hudson-Raritan Estuary were monitored over the past decade, and limited monitoring of the Bight did not begin until 1976. Also, research measurements during the 1970s were seldom conducted over time periods long enough to detect possible trends. However, a few ecosystem features are known well enough to document the existence or probable absence of significant decadal trends. Changes are evident in the community structure and commercial catch of demersal fishes and shellfish (Grosslein and Azarovitz, 1981). However, no major trends are noticeable in bacterial measures of water quality nor in the locations of contaminated sediments of the inner Bight. It would not be surprising if additional trends in contaminant loadings and ecosystem impacts did occur during the 1970s but went undetected.

Bathing Water Quality—In the New York and New Jersey coastal zone, swimming is the most popular outdoor recreation (Carls, 1978). The City of New York has increased its public beachfront from 1.6 km (1 mi) in 1933 to 29 km (18 mi). Increasing sewage loadings to the New York Harbor since the 1800s have caused widespread closures of bathing beaches (Suszkowski, 1973). Despite continuing efforts to manage sewage wastes since the early 1900s, the entire inner harbor is not classified as acceptable for bathing. Of the 12 recognized beaches in Lower Bay, most were open to bathing, with intermittent closures, during the late 1970s (New York City, 1979). There is no evidence of serious disease associated with swimming in the harbor since 1920, but upper respiratory inflammations and gastroenteritis have been associated with swimming in recent years. The bacterial or viral agents of these diseases are not yet identified, but the probability of illness among swimmers can be predicted rather reliably from concentrations of appropriate indicator bacteria in bathing waters. Based upon epidemiological observations at beaches of Coney Island and western Long Island, *Enterococcus* and *Escherichia coli* were the best indicators of swimming-associated gastrointestinal symptoms (vomiting, diarrhea, nausea, or stomachache). The probability of contracting such symptoms was generally higher at the Coney Island than at the Long Island beaches. These studies also documented an appreciable incidence of gastroenteritis from swimming in approved bathing waters (Cabelli, 1981).

There is no evidence that municipal waste discharges from Long Island have diminished water quality beyond existing bathing standards. Municipal wastewaters from New York City and the New Jersey coast seldom cause coliform densities to exceed bathing water standards, at least along the open coast (Cabelli, 1981).

Although bacterial and other water quality data for the 1970s over the Bight region are incomplete and not exhaustively analyzed, there seem to have been no striking wide-scale trends in bathing water quality since 1970 (Gunnerson, 1981; Cabelli, 1981).

Shellfish Sanitation—Concentrations of total and fecal coliform bacteria, indicators of human pathogens, have been high enough for several years to require closure of most Hudson-Raritan Estuary waters to shellfish harvesting. Even the New York Harbor waters classified for shellfishing use (Raritan and Sandy Hook Bays) have not met the New York State or U.S. Environmental Protection Agency coliform standards for shellfishing in recent years (New York City, 1979). Similar closures were instituted in the apex in 1970 and extended geographically in 1974 (Verber, 1976). Measurements in the water column of the apex indicate that coliform bacterial concentrations from sewage sludge dumping reach background levels within 3 to 5 km (2 to 3 mi) of the barge discharge site, whereas elevated concentrations in the estuarine plume extend much farther into the apex (O'Connor, D.J., et al., 1977).

Petroleum Pollution—Petroleum hydrocarbons have been identified as a class of contaminants of concern within the Bight region. Of particular concern are the polynuclear aromatics (PNAHs) such as the benzenes, naphthalene, and benz-anthracenes. Because of their toxicity, carcinogenicity, and relatively high concentrations in the ecosystem, the PNAHs have been characterized as "major perceived threats that require continued study" in the Bight region (O'Connor, J.S., and Stanford, 1979). Petroleum hydrocarbons have also formed surface slicks, have fouled beaches, and have tainted fish and shellfish.

The daily chronic input of oil and grease to the Bight region has been estimated at 870 t/day (Mueller et al., 1976). Assuming that 60 percent of this material constitutes petroleum hydrocarbons (NAS, 1975) with an average density of 0.95, the average daily oil input would be about 520 t or 550 m^3/d (0.15 mgd). This estimate does not include atmospheric inputs from fossil fuel burning that have been estimated at 17 to 42 m^3/d (Gibson et al., 1979). Thus, total chronic petroleum hydrocarbon loadings are estimated at 570 to 590 m^3/d (0.15 to 0.16 mgd). The quantities lost from routine ship operations are unknown.

Individual oil spills can release large quantities of oil in small regions, resulting in major impacts. However, the average quantities released from spills appear to be much less than those from chronic oil losses to the Bight region. The U.S. Coast Guard Pollution Incident Reporting System (PIRS) records that all known spills of petroleum hydrocarbons from 1974 through 1979 have averaged 0.01 m^3/d (0.003 mgd), i.e., much less than 1 percent of all petroleum hydrocarbons introduced to the Hudson-Raritan Estuary and New York Bight.

Despite the substantial dispersal of petroleum hydrocarbons by dissolution, evaporation, and degradation, large quantities are found in the Bight region. A large proportion of the introduced petroleum hydrocarbons reach the sediments. Sediments with high hydrocarbon concentrations are particularly evident in the Hudson-Raritan Estuary. Concentrations of PNAHs alone in sediments of the estuary range from 3 to 180 $\mu g/g$ dry wt. The hydrocarbons in particularly high concentrations are naphthalene, phenanthrene, and fluoranthene (Anderson, 1981). Concentrations of all petroleum hydrocarbons in sediments of the Bight range from 500 to 3,000 $\mu g/g$ dry wt. in fine sediments of the apex to about 10 $\mu g/g$ dry wt. on the continental shelf (Farrington and Tripp, 1979).

The organisms analyzed appear to accumulate naphthalene and biphenyl more than the other PNAHs. Digestive glands of lobsters accumulate higher concentrations and a broader spectrum of PNAHs than any of nine other species analyzed (MacLeod et al., in press). The degree of hydrocarbon tainting in food species and chronic effects on biota require further analysis (Anderson, 1981).

The above estimates are based upon data gathered throughout the 1970s. Available data do not permit any reliable assessment of trends in petroleum hydrocarbon input rates or accumulation in the ecosystem over the past 10 years.

Dredged Material—The large quantities of natural riverborne sediments and anthropogenic particulate inputs are rather effectively trapped in New York Harbor. These sediments tend to accumulate in the 386 km (240 mi) of federally maintained channels and commercial/recreational channels required for vessel traffic in the harbor. Dredging of these channels has removed an average of about 8 million m^3/yr (10 million yd^3/yr) from the harbor during the 1970s and only slightly less per year since 1930. During the 1970s most of the dredged material has been disposed of at the "mud dump" site in the apex.

Because most toxicants introduced to harbor waters tend to adhere to particles and settle to bottom sediments, concern has long been expressed over the disturbance and ocean dumping of contaminated sediments. The degree of dredged material contamination varies greatly. One assessment indicates that about 10 percent of the material dredged from the harbor is clean sand and at least 10 percent is highly contaminated and cannot be dumped into the ocean under existing regulatory criteria (Gordon et al., 1981).

For several years, the wide variety of organic and inorganic toxicants in dredged materials has stimulated concern about effects upon the ecosystem of the Bight and toxicant accumulations in marine food resources. This concern has heightened since 1976, when large quantities of PCBs were identified in sediments of the Hudson River. From 200,000 to 300,000 kg (440,000 to 660,000 lb) of PCBs remain in the sediments of the Hudson from discharges by capacitor manufacturing plants about 400 km (250 mi) upstream from Manhattan. Although the rate of transport of this PCB reservoir to the harbor has not been estimated, there is a high probability that large proportions are being carried to the harbor (O'Connor, J.M., et al., 1981). While additional measurements of PCB inputs to the Bight will be required for reliable estimates, it appears that dredged materials are already a major source of PCBs to the Bight and that these materials may accumulate significantly higher PCB concentrations than existing ones from the Hudson River (O'Connor, J.M., et al., 1981).

Environmental Crisis—Real or Imaginary

At the outset of the 1970s, public awareness of environmental problems was increasing rapidly and an intense review was taking place aimed at identifying and eliminating the unacceptable or unnecessary impacts of human activities on the environment. During this period attention was focused most sharply on the more immediately visible sources of pollution such as the automobile and its smog-producing ability and floatable materials from sewage discharges. In the New York region, attention became focused on the large quantities of sewage sludge barged out to sea and dumped in the ocean. During the seventies several environmental crises occurred in the New York Bight, each of which was linked in its own way with the ocean dumping issue.

Sewage Sludge: Beach Pollution and Anoxia—Public concern over the impacts of ocean dumping in the New York Bight grew out of a series of observations of environmental damage. The most important of these observations was the detection of high concentrations of coliform bacteria in waters near the dredged material and sewage sludge ocean dumpsites (Buelow et al., 1968) as a direct consequence of which an area of radius 11 km (7 mi) around the sewage sludge dumpsite was closed to shellfishing in May 1970. This was apparently the first instance of shellfish habitat closure on any open U.S. continental shelf. At the same time, the Congress was considering the need for legislation to regulate and control ocean dumping and dumping in the Great Lakes. Media accounts prompted by the early stage of the Congressional consideration of this legislation contained reports that the dumping had created a "dead sea" in the Bight, that the contaminated area, then 50 km^2 (20 mi^2), was "growing rapidly," and that this could necessitate closing New York City area beaches in the coming (1970) summer (Madden, 1970). All these contentions were unsupported by the meager scientific information then available and were seriously misleading. However, the public accounts did detail the indications of environmental degradation that did exist, including evidence of depauperate benthic fauna, diseased fish and shellfish, unusually high coliform bacterial counts in Bight waters and sediments, and high metal concentrations in the sediments.

Later in 1970 the Council on Environmental Quality (CEQ) published a report on ocean dumping. This report reviewed the limited scientific information available at that time, much of which was contained in unreviewed technical reports, and concluded that ocean dumping was "not a serious, nationwide problem," but that "in some areas the environmental conditions created by the ocean disposal of wastes are serious" (CEQ, 1970). This latter conclusion was substantially qualified in the report itself by the statement that "knowledge of ocean pollution is rudimentary, and generally it has not been possible to separate the effects of ocean dumping from the broader issue of ocean pollution (CEQ, 1970). This statement is particularly relevant to the Bight because there are so many sources of pollution. Despite the considerable acknowledged uncertainty and lack of adequate data, the CEQ report made strong recommendations that ocean dumping should be subject to regulation and that ocean dumping of sewage sludge and polluted dredged material should be phased out (CEQ, 1970).

In response primarily to the call of the CEQ report for strong national legislation to regulate ocean dumping, the Congress enacted the Marine Protection Research and Sanctuaries Act, which became law in October of 1972 (33 USC S1401 *et seq*). The Act states that:

> The Congress declares that it is the policy of the United States to regulate the dumping of all types of materials into ocean waters and to prevent or strictly limit the dumping into ocean waters of any material which would adversely affect human health, welfare or amenities or the marine environment ecological systems or economic potentialities. (PL-95-532, 33 USC S1401)

Immediately before and after the passage of the Act a number of research programs were initiated to investigate more fully the effects of ocean dumping in the New York Bight. In December 1973 and early in 1974, the popular press obtained preliminary unpublished observations from limited sampling that constituted the early results of such studies. Press accounts that followed included references to: 1) a "dead sea" from the sewage sludge dumping (Sharov, 1973), 2) migration of the sludge "bed" to within one-half mile of Long Island bathing beaches (Bird, 1973), 3) predictions of the sludge would soon begin to wash up on the beaches (Bird, 1973; Pearson, 1974; Carroll, 1974), and 4) warnings about potentially serious public health hazards from heavy metals, bacteria, and viruses (Carroll, 1974; Kline, 1974; Pearson, 1974).

Other accounts claimed that the existing sewage sludge dumpsite could be used for only one more year because the "dead sea" created by sludge was moving toward Long Island beaches. This ominous prediction for onshore displacement of the sludge bed gained credibility because it was attributed to "an authoritative EPA source" (Pearson, 1974), that is, to a responsible regulatory agency. During this period of time, stories and predictions of environmental disaster such as those cited above appeared in national magazines and more than 100 newspapers, including newspapers from the West Coast, and were heard on radio and television (Kidder, 1975; Souci, 1974). In addition the U.S. Environmental Protection Agency itself issued a report stating: "We clearly recognize that the practice [of ocean sewage sludge dumping] over the past 45 years has created a dead sea in the general area of this site" (U.S. Environmental Protection Agency Briefing Report, 1974).

In response to the public and political pressures to take some action concerning the newspaper media reports of impending environmental disaster caused by continued sewage sludge dumping, the U.S. Environmental Protection Agency in 1974 notified the municipalities responsible for dumping sewage sludge that they would be expected to use an alternative offshore sludge dumping site within 2 years. During the months following this announcement, the Environmental Protection Agency evaluated the available scientific information concerning sludge dumping in the ocean in order to develop an environmental impact statement in which moving the existing sewage sludge dumpsite to alternate sites further offshore was considered.

The draft of this environmental impact statement (EIS) was issued in February of 1976 (U.S. Environmental Protection Agency, EIS, Draft, 1976; Final Report, 1978). The conclusions of the draft, which remained unchanged in the Final EIS issued in September 1978, include the following:

- Sewage sludge dumping at the existing site has not significantly affected the water quality off Long Island or New Jersey beaches.
- Dumping of current volumes of sewage sludge will not have a significant effect on the rather limited benthic community at the existing site. The benthic community would not recover in the near future if the existing site were abandoned. Furthermore, areas now closed to shellfishing would not be reopened in the near future, even if the existing site were abandoned.
- Continued dumping of present volumes of sewage sludge at the existing site will not have a significant additional effect on the water quality in the Bight apex.
- Small quantities of floatables derived from sewage sludge are present at the existing dumpsite for short periods immediately after dumping occurs. There is no direct evidence that the wash-up of floatables on Long Island and New Jersey is attributable to sewage sludge dumping. The probability that these materials result directly from sludge dumping activities is low.

These conclusions led the Environmental Protection Agency to recommend that the existing sewage sludge dumpsite should continue to be used, although they believed that the development and implementation of land-based alternatives that are environmentally acceptable, technically feasible, and economically reasonable should be carried forth as expeditiously as possible. At the same time, this draft environmental impact statement contained references to several ongoing and completed studies that indicated sewage sludge was only a minor contributor to the overall pollution problem within the New York Bight apex (Mueller et al., 1976; MESA, 1975; Segar and Cantillo, 1975; Segar et al., 1975; Drake, 1974).

In June 1976 almost all of Long Island's major public ocean beaches were closed to swimmers for varying periods of time because of floating trash and pollutants. Waterborne debris has been a constant irritant to beach users in recent years, but the concentrations during June 1976 were the heaviest ever known. The unprecedented closings began with the restriction of 32 km (20 mi) of Fire Island beaches on June 15, 1976. By the third week of June 1976, most of Long Island's south shore beaches were closed. By July 1, 1976, these beaches were again opened, but during the interval, normal summer beach use decreased, causing inconvenience and annoyance to prospective swimmers and economic loss to local businesses. The problem was such that on June 23, 1976, the area was declared a disaster area by the Governor of the State of New York. Because much of the material washing up on the beaches appeared to be derived from sewage, there was public suspicion that the source of the material was the sewage sludge dumped into the ocean off the shore of Long Island. This suspicion persisted despite the Environmental Protection Agency's finding in the environmental impact statement that floatables from the sewage sludge dumping were negligible in quantity. In February 1977, a detailed analysis of the Long Island beach pollution incident was reported by the National Oceanic and Atmospheric Administration (MESA, 1977). This report concluded that no source could be identified as the single major contributor of floatables. However, the report continued, most of the material was probably derived from the outflow of the Hudson-Raritan Estuary and, although sewage sludge dumping was a possible source of floatables, the contribution from it was "relatively minor."

In July 1976 fishermen reported large numbers of dead surf clams and other bottom-dwelling organisms in an 8,600 km^2 (3,320 mi^2) area off the New Jersey continental shelf. The phenomenon continued through October of that year. The mortalities were caused by extremely low concentrations of dissolved oxygen and by hydrogen sulfide poisoning in some bottom waters. At the height of the event, dissolved oxygen values in the water approached and in some instances reached zero in an area lying 10 to 100 km (6 to 62 mi) off the coast between Sandy Hook and Cape May. Mortalities were greatest among surf clams, ocean quahogs, and other benthic animals. Lobster catches declined almost 50 percent during the period. These events have been described in detail by Swanson and Sindermann (1979). As a result of this crisis, in November 1976, the federal government declared the New Jersey coast a resource disaster area. Estimates of losses to the commercial and recreational fishing industries and related processing and service businesses were as high as $550 million. Local fishermen were also concerned about the long-term impact of this event on their fisheries. Despite the fact that the sewage sludge dumping was known to contribute only a small proportion of the oxygen demand within the New York Bight apex (Segar and Berberian, 1976), once again sewage sludge dumping became the object of suspicion among the public.

Here then in 1976 in the floatables incident and the oxygen depletion event were two environmental disasters of just the nature that had been predicted in the early 1970s. What could be more natural than for the public to conclude that the earlier investigators had been correct and that the sewage sludge dumping was indeed responsible for these two environmental events? Public pressure for government action to prevent the happenings of 1976 from recurring was extremely strong. The real causes of these two events were floatables entering the rivers from diverse sources in the New York region (MESA, 1977) and natural changes in the physical and biological characteristics of the waters of the New York Bight, augmented by nutrient inputs from the estuary and ocean outfalls and to a lesser extent from ocean dumping (Swanson and Sindermann, 1979). Therefore, the Congress reacted in 1977 by enacting an amendment to the Marine Protection Research and Sanctuaries Act that established a mandatory deadline of December 31, 1981, for the termination of "harmful" sewage sludge dumping in the ocean. This Congressional action was based largely upon public misconceptions rather than scientific fact and did not consider the impacts of land-based alternatives as fully as ocean dumping alternatives. The deadline established by the 1977 amendment was not absolute but applied only to sewage sludge that would "unreasonably degrade or endanger human health, welfare, amenities or the marine environmental ecological systems or economic potentialities" (PL 95-153, 33 USC S1401). Despite this clear statement by the Congress that dumping of some sewage sludges into the ocean after 1981 was acceptable, provided that unreasonable degradation did not occur, the amendment has been consistently misinterpreted in the public arena as an absolute ban on all dumping of all sewage sludges.

As we have described, during the period between 1970 and 1976 two real environmental crises preceded by one imaginary environmental crisis occurred in the New York Bight. Sewage sludge is an inherently aesthetically displeasing substance to our society. Therefore, it is not surprising that the media were able to generate considerable public concern when it appeared likely that the sewage sludge would affect Long Island beaches. The technical information gathered and reviewed when the moving of the dumpsites was considered indicated that sewage sludge dumping in the New York Bight apex contributed only a minor portion of the contaminant inputs causing environmental degradation and the potential for environmental crises (U.S. Environmental Protection Agency, 1978). The institutional response to the 1976 oxygen depletion and beach pollution events did not take into account this technical information, since the single governmental action was to establish a statutory deadline for phasing out ocean dumping of sewage sludge. While stoppage of sewage sludge dumping will diminish inshore eutrophication minimally and reduce toxicant loadings to some extent, the other anthropogenic sources dominate impacts upon the Bight. There has been no comparable regulatory action to minimize the discharge of floatable materials or toxicants to the estuary and Bight.

Dredged Material/Bioaccumulation—The waters in New York Harbor are naturally shallow, and dredging is required to maintain channels deep enough for the safe navigation of ships. For about 200 years, open water disposal sites located near the entrance to New York Harbor have been utilized to receive materials from dredging of the harbor's access channels and ship berths. The disposal sites have been moved several times, the present disposal grounds in the New York Bight being approximately 10 km (6 mi) east of Highlands, New Jersey, and 16 km (10 mi) south of Rockaway Beach (Gross, 1976). Dredged material consists of natural material originating in the watersheds of the Hudson and other rivers entering the harbor; solid material entering the waterways through sewage treatment plant discharges, storm sewers, and other outfalls; and material brought in with tidal flow from the Atlantic Ocean. Dredged material varies from clean sand with very low organic content and extremely low concentrations of trace metals and synthetic organic contaminants to contaminated sediments containing several percent of organic matter and high concentrations of trace metals and synthetic organic compounds. The ongoing issue of the environmental impact of disposal of large quantities of dredged material with associated quantities of toxic metals and synthetic organics has been discussed above. However, ocean dumping of polluted dredged material has created a recent environmental "crisis." Although the "crisis" addressed only a portion of the overall problem, it focused public attention on this issue.

The dumping of dredged material in the ocean has been regulated under the Marine Protection Research and Sanctuaries Act since 1972. In view of the lack of understanding of the environmental impact of solid wastes disposed in the ocean, the regulations pursuant to the Marine Protection Research and Sanctuaries Act were amended in January 1977, so that criteria for determining whether a material is suitable for ocean disposal are based on the use of bioassay techniques and bioaccumulation tests (40 CFR Part 227; 42 *Fed Reg* 2476-89, January 11, 1977). Implementation of the bioassay procedures began early in 1978. All dredged materials from the New York area were found to pass the new bioassay-based criteria as these were interpreted by the Corps of Engineers, although significant questions have been raised concerning the Corps' interpretation of the criteria. This issue is the subject of an ongoing lawsuit brought by the National Wildlife Federation (Kamlet, 1981). Bioaccumulation testing was required after February 1979. When bioaccumulation data began to become available for dredged material, it became apparent that, at least with respect to PCBs, many contaminated dredged materials could not pass the bioaccumulation criteria as interpreted by the Environmental Protection Agency and that under existing guidelines, no permit for ocean disposal could be issued for these dredged materials.

This situation raised the serious question of whether parts of the Port of New York and New Jersey would be forced to close, since without an ocean disposal permit, there was no reasonable means of disposing of material resulting from maintenance dredging of essential ship channels. The potential economic and social disruption that would have been caused by such a closure was sufficient to generate considerable public concern, and the issue of dredged material disposal in the ocean became an environmental "crisis." During the early months of 1979, several permit applications for maintenance dredging in the Port of New York and New Jersey were suspended while the Corps of Engineers and Environmental Protection Agency tried to decide whether or not these permits could be issued in view of the positive findings of the bioaccumulation tests for PCBs. Channel and berth siltation, meanwhile, continued, and the availability of adequate berths was in doubt for the liners *Queen Elizabeth II*, *Rotterdam*, and *Norway* due to arrive in April and May. The necessary dredging permits were finally issued in March.

Under the ocean dumping regulations, certain specified contaminants including PCBs can be ocean dumped only when present in ocean dumped materials "in such forms and amounts . . . that the dumping of the materials will not cause significant undesirable effects, including the possibility of danger associated with the bioaccumulation in marine organisms" (40 CFR 227.6b). The Environmental Protection Agency/Corps of Engineers' implementation manual that specifies the bioaccumulation test procedures states that "in order to ensure environmental safety, it must be assumed that any statistically significant bioaccumulation relative to animals not in dredged material but living in material of similar sedimentological character, is potentially undesirable" (U.S. Environmental Protection Agency/Corps of Engineers, 1977). The manual further recommends "the environmentally protective approach of assuming that any statistically significant differences in tissue concentrations between control and exposed organisms are a potential cause for concern." However, noting that at present "tissue concentrations of most constituents in most species cannot be quantitatively related to biological effects," the manual calls upon the Environmental Protection Agency and the Corps of Engineers to "objectively consider the magnitude of bioaccumulation shown, the toxicological significance of the material bioaccumulated, the proportion of sediment sampling sites which produce uptake, the number of different constituents bioaccumulated from the sediments in question, the position in the human and nonhuman food webs of the species showing uptake, the presence of motile species at the site which might serve as transportation vectors removing bioaccumulated materials from the disposal area, and other factors relative to the particular operation in question."

In a January 9, 1979, letter to the New York District of the Corps of Engineers, Environmental Protection Agency Region II adopted the following position on interpretation of bioaccumulation test results: "In view of existing FDA criteria limiting the parameters to be tested in the bioaccumulation studies and thereby identifying them as potential threats to public health and welfare, and consistent with the intent of Section 226.6 [c] of the regulations and the COE/EPA Manual, paragraph G32, *any* statistically significant bioaccumulation would be considered cause for denial, unless such statistically significant difference is shown to have no significant adverse effect on public health and welfare."

Early in 1980, the Corps of Engineers and Environmental Protection Agency formed a joint task force that was charged with preparing a matrix for developing more interpretive guidelines to evaluate the PCB problem. Meanwhile, the various dredging permits concerned were being held in abeyance. The task force developed an interpretive matrix under which dredged materials could be considered for approval for ocean dumping if a statistically significant increase of PCB concentrations occurred in a bioaccumulation test. This bioaccumulation test used three organisms: a worm, a clam, and a shrimp. If statistically significant increases occur in all three test organisms, then the material is not in compliance with the criteria and, therefore, cannot be ocean dumped. If a statistically significant increase occurs in at most two of the test organisms, the material is considered in compliance with the criteria and can be ocean dumped without constraint, provided that the level of bioaccumulation (final tissue concentration observed) in both of the organisms showing statistically significant increase is below specified threshold values. If a statistically significant increase occurs in at most two of the test organisms and the magnitude of the uptake exceeds these threshold values, the material is considered to be unsuitable for unconstrained ocean dumping. On a case by case basis, the acceptability of ocean dumping this material under certain circumstances (for example, when it is capped by clean material) remains open.

This rather complex matrix is an expression of the difficulty in interpreting bioaccumulation data, because information concerning the ecological significance of bioaccumulation of toxic components in laboratory organisms is scarce. This is a limitation inherent in applying any laboratory bioassay or bioaccumulation test to the determination of the potential for environmental impact. While the development and use of such laboratory tests is less expensive than field data collection and analysis, the use of such tests alone, as exemplified by the dredged material problem, is often unsatisfactory and leads to poor decision making.

At present the dumping of dredged material in the ocean is still controlled by criteria requiring bioaccumulation tests and a matrix approach to interpreting the results of those tests. The Environmental Protection Agency and the Corps of Engineers have stated that this matrix is only to be used on an interim basis and that attempts will be made within the immediate future to review the ocean dumping criteria and to develop new criteria that might more accurately reflect the potential for environmental degradation. Such revised criteria should take into account the need for an assessment of ecological effects in the region of the dumpsite itself either to replace the laboratory bioassays and bioaccumulation tests or to supplement these laboratory results and facilitate their interpretation. Criteria revised in this manner would not only allow the current bioaccumulation test controversy to be resolved but would also aid in better identifying the nature and extent of the overall environmental impact of dredged material ocean disposal.

LIMITATIONS OF STATUTES AND REGULATIONS

Historically, the seventies will be viewed as the decade of environmental legislation. The decade began with the signing into law of the National Environmental Policy Act [NEPA] on January 3, 1970. After NEPA came the Clean Air Act of 1970 and then, in the 92d Congress, six new statutes came into being: the Federal Water Pollution and Control Act; the Federal Insecticide, Fungicide and Rodenticide Act; the Marine Mammal Protection Act; the Marine Protection Research and Sanctuaries Act; the Noise Control Act; and the Coastal Zone Management Act. The 93d. Congress passed the Endangered Species Act, the Safe Drinking Water Act, and the Deep Water Port Act. The 94th Congress enacted the Toxic Substance Control Act and the Resource Conservation and Recovery Act. The 95th Congress passed major amendments to the Clean Air Act, the Federal Water Pollution Control Act, the Safe Drinking Water Act, and the Outer Continental Shelf Lands Act.

One major drawback to the piecemeal approach taken to environmental protection in the 1970s is that it resulted in a disjointed management of the environment by medium, rather than an integrated approach to dealing with environmental problems. Regulations promulgated pursuant to the various environmental laws each adopt substantially different approaches to determining whether discharge or disposal of a material is acceptable under each given Act. As a result instead of comparing waste disposal options in different media and selecting the optimal option on environmental, social, and economic grounds, an industry or a municipality faced with the need to dispose of its wastes will often seek to find the option that is least stringently regulated. One particularly notable example of this problem relates to the disposal of dredged material, which is regulated under the Federal Water Pollution Control Act for inland waters and the territorial sea (40 CFR S230), and under the Marine Protection, Research, and Sanctuaries Act for the territorial sea and contiguous ocean (40 CFR S220-229). The regulations for ocean disposal are significantly more stringent than those for inland water disposal, thus encouraging inland water disposal, which may be more harmful than ocean disposal. Aside from the problems caused by fragmentation of statutes and regulations, additional problems occur because of inexperience with environmental law and regulations and their operation. Ten years is simply too short a time for a body of case law to be laid down sufficient to identify all the problems with existing legislative and regulatory approaches let alone to permit amendment and modification of the approaches such that they result in efficient application.

Our ability to efficiently and effectively manage and control pollution of the environment in general and the ocean environment in particular is constrained by the lack of conformity among the various environmental statutes and also by shortcomings in the regulatory framework built around those statutes. During the next decade it should be our aim to learn to apply the existing statutes, to modify them so as to bring a degree of uniformity and efficiency into their operation, and, as an outcome, to produce optimal solutions to environmental problems. The areas where such amendments to statutes or to regulations will be needed are numerous. It is instructive to review a few of these areas as they apply to problems within the New York Bight.

NPDES and SPDES Inadequacies

The New York Bight is contaminated predominantly by discharges either directly into the ocean or into the river and estuary system through pipelines. The contaminants enter the New York Bight directly through ocean outfalls, or after they are discharged through an estuarine or river outfall and carried out to sea by the estuarine outflow. They may also enter the Bight as a component of dredged material taken from the channels within the estuarine system. Pipeline discharges to the ocean or river and estuarine system are regulated under the Clean Water Act and are subject to the permit procedures of this Act, known as the National Pollutant Discharge Elimination System (NPDES) or the State Pollutant Discharge Elimination System (SPDES), where a state assumes permitting responsibility under the terms of the Act.

The NPDES program relies on the direct control of waste discharges through a series of effluent concentration standards in order to achieve the desired ambient water quality characteristics. Central among the issues concerned with the NPDES strategy and its implementation is the effectiveness of relying on uniform national or regional effluent standards. The adoption of uniform effluent standards is advantageous, particularly given the ease of the negotiation process between government and a discharger (Energy and Environmental Analysis, Inc., 1975). However, the adoption of uniform national standards has led inevitably to numerous cases in which variances from these standards may be justifiable because of local environmental conditions or constraints, or economic considerations, but no adequate mechanism for granting such variances exists. At present any variances must be granted through statutory exemption (Blumm, 1980). Such variances have been granted on a class by class basis rather than on an individual discharger basis. The formal variances that currently exist are those for power plants (Section 316 [a] of the Clean Water Act of 1977) and exemptions of municipal wastewater treatment plant discharges from the secondary treatment requirement if discharge is through an ocean outfall where a large amount of dilution is probable (Section 301 [h] of the Clean Water Act of 1977). It is likely that more variances will be requested as other parts of the Clean Water Act, such as toxic substances controls, are implemented (Blumm, 1980) and as more information concerning local environmental conditions becomes available. If these variances from the national standards do in fact proliferate, particularly if they are justified largely by local water quality conditions, then it has been suggested that this would constitute a de facto movement back to standards based on water quality and water use as opposed to the current technology-based standards (Blumm, 1980).

One shortcoming of the NPDES system is the problem of monitoring compliance with permit conditions. Little hard information is available to enable the dimensions of this problem to be adequately identified. However, it appears likely that because of the self-monitoring aspects of the permit procedures and the limited data that the permit procedures require, significant violations of permit conditions may take place without detection.

One of the major difficulties in applying the current national effluent standards approach is that the standards are established on an industry by industry basis and, where combinations of discharges occur or where unique plant or process streams are concerned, the effluent stream may not fit into one of the categories for which standards have been promulgated. In this case the effluent discharge limitations included in the permit must be designed specifically for the particular discharger. This situation leads to the possibility that a particular industry or process stream can obtain a de facto variance from the effluent standards by establishing that the process stream or plant concerned is unique and that different effluent standards should then be written for it.

A major inadequacy in the current NPDES program is the limitation of the NPDES system in controlling toxic substances. Prior to the 1977 Clean Water Act, the NPDES program and its predecessor concentrated primarily on conventional pollutants, though the permit applications often listed all of the characteristics of wastewater discharges including toxic substances. While the 1972 Act did require toxic substance standards to be developed and implemented through the NPDES system, this plan never came to fruition for reasons related to the complexity of setting toxic standards and industry resistance (Blumm, 1980). The Clean Water Act has intensified the emphasis of the statute on toxic substance standards. Instead of adopting widely applicable toxic effluent standards, implementation of regulations concerning these substances in waste process streams will take place through technology-based standards for NPDES discharges, pretreatment standards for discharges through publicly owned waste treatment plants (where toxic substances prove to be incompatible with plant operation), and specific standards for new sources (Blumm, 1980). It is not clear that this essentially complex system will be successful in substantially reducing the load of toxic substances entering the waterways of the New York region and thereby entering the New York Bight.

Nonpoint Sources of Contaminants

Although the many existing statutes of environmental law control the placement of waste materials in all media including the oceans, the atmosphere, the land and underground water tables, and the release of contaminants from specific point sources such as industrial plants to the atmosphere and the water and land environment, a multitude of nonpoint sources of contaminants to the environment still exist that are not adequately regulated. In a coastal ecosystem such as the New York Bight, these nonpoint sources are limited to atmospheric fallout and precipitation, erosion of and runoff from the land, together with the many minor events of uncontrolled and undocumented disposal of wastes by the public or small businesses, and discharges from the many vessels utilizing the New York Bight. The quantity of contaminants introduced to the ocean by these nonpoint sources can in some instances be quite large. Duce et al. (1976), for example, have estimated that up to 13 percent of the lead, 8 percent of the zinc, 5 percent of the iron, and 1 to 2 percent of the cadmium entering the New York Bight may do so by way of the atmosphere through particulate fallout and rainfall. Kneip et al. (1981) have similarly estimated that the contribution of PCBs through atmospheric fallout to the New York Bight is significant compared to other sources.

The most important nonpoint source of contaminants to the estuarine and ocean system appears to be the runoff of storm water, washing contaminant-laden material off the streets and the land. Street and land runoff contain diverse contaminants such as agricultural chemicals, hydrocarbons from crankcase and other waste oils, and synthetic organics including PCBs and trace metals from a multitude of diverse sources. In urban areas such as the New York-New Jersey region, where sewer systems and storm drain systems are combined, much of the material from nonpoint sources entering the water environment does so through combined sewer overflows during periods of rainfall, as discussed in more detail below.

Although the nonpoint sources of contaminants entering the New York Bight are at present small compared to the direct sources, they will constitute a growing proportion of the total contaminant load as the concentrations or loadings of toxic contaminants in direct sources are brought under control by the various environmental statutes. For at least some contaminants, it is certain to prove necessary or desirable to reduce contaminant loadings to levels that will require control of the nonpoint sources. This is particularly likely for petroleum hydrocarbons. At present, the existing environmental statutes do not adequately address the need or the means to reduce such nonpoint sources and the technological problems that exist with developing such legislation and control practices are difficult. The lack of regulations to control nonpoint sources of contaminants is a potentially serious environmental problem. The development of such regulations and technologies whereby the regulations can be implemented should be a matter of priority.

Treatment Plant Inadequacies

Although almost all municipal and industrial waste discharges into the aquatic environment are now regulated, large quantities of contaminants are still contained in the permitted waste discharges owing to operational and maintenance inadequacies of treatment plants that restrict the degree to which permitted discharge rates can be maintained. Over 70 percent of the sewered areas in the New York metropolitan region have combined sewers. During dry weather they function as sanitary sewers, conveying all flows to the treatment plants. During wet weather, large volumes of rainfall runoff enter the system—the average storm triples the normal dry weather flow, but peak flow can be as much as 50 times the normal flow. Waterfront regulators are built into the sewage systems to act as relief valves to prevent flooding of treatment plants during wet weather. These regulators allow no more than twice the average dry weather flow to reach the plants so that even during the average storm a large proportion of the combined flow is simply discharged through the regulators without treatment. In addition many of the regulators are not in good operating condition and leak during dry weather.

Table 1 shows the average daily quantities of various contaminants released by New York and New Jersey municipal wastewater treatment works. The New York City raw bypass is untreated sewage discharged where no treatment plant has yet been constructed and where several treatment plants were closed for construction. The noncontrolled discharges (regulator leakage, combined sewer overflows, and storm runoff) contribute a large proportion of the total amount of the various contaminants released. Although there are plans to upgrade and repair regulators and to partially treat combined sewer overflows, it is unlikely that these inputs can be substantially reduced and maintained at a low level unless major technological progress is made or major new treatment capabilities constructed.

For comparative purposes, the total quantities of the same contaminants in sewage sludge generated by the New York-New Jersey treatment plants are also included in Table 1. The quantity of the metals released through the effluents of operating treatment plants, not even considering storm runoff, combined sewer overflows, and regulator leakage, far exceeds the quantity retained in the sewage sludge and barged for ocean dumping. This is an important point, since at present the laws and regulations governing wastewater treatment and disposal emphasize more strongly the elimination of ocean sewage sludge dumping than control of the other treatment plant discharges and storm runoff. This may not be the optimal strategy for achieving the maximum immediate environmental benefit through use of the limited funds available for environmental improvement.

The general areas of regulatory shortcomings briefly discussed above are only three of many. The process of amending our young body of environmental laws and regulations into a coherent and effective whole will take several years, but further efforts should be made in the interim either to enforce or rescind existing regulations not being implemented.

In the realm of regulatory techniques or strategies, it is clear that there are constraints on monitoring compliance, particularly when regulating the large numbers of small sources or nonpoint sources not now regulated or ineffectively regulated. In recognition of these constraints, a system of incentives must be developed to ensure that the discharger seeks to maintain good management practices, not simply to avoid possible detection and prosecution of violations that the discharger knows are unlikely but out of self interest. Designing such a scheme of incentives may well be the greatest environmental challenge of the 1980s.

One of the most vital scientific information needs for improved environmental management is the determination of the capacity of natural ecosystems to assimilate, or otherwise cope with, various wastes and waste components (Goldberg, 1979). Such knowledge is essential not only to enable limits to be placed on the quantities of contaminants released to the environment but also to allow for the management of the releases themselves in the most environmentally sound and cost effective manner.

Table 1. Sources of Pollutants to Hudson-Raritan Estuary and Metals in Sewage Sludge*

	BOD (1000 lbs/d)	Susp. solids (1000 lbs/d)	Total Kjeldahl nitrogen (100 lbs/d)	Fecal coliform (MPN/d)	Metals (lbs/d)				
					Cu	Cr	Ni	Zn	Pb
Sewage plant effluent	711	905	2622	3×10^{17}	2700	2600	2200	8300	1000
Regulator leakage	35	81	48	6.6×10^{15}	56	41	53	110	32
Combined sewer overflow	182	275	301	1.7×10^{16}	390	170	170	670	420
Storm runoff	95	574	114	1.2×10^{15}	1600	840	490	2100	1900
Raw sewage bypass	524	505	846	1.2×10^{17}	910	540	690	1100	290
Sewage sludge	440†	990†	37†	3.6×10^{13}†	780	370	90	900	770

*1975 data from New York City 208 Study, 1978. Task 516/526: Baseline/Alternatives Summary, Tables 1-1 through 1-38. Prepared by Managing Consultants, Hazen and Sawyer, New York, NY.

†From Mueller, et al., 1976, Table 11.

THE LAST TEN YEARS

The past 10 years have seen a dramatic awakening of public and institutional interest in the effects of our society on the environment, including the oceans. Nowhere has this awakening been more clearly felt than in the New York-New Jersey metropolitan region.

The problems of the New York Bight are but one component of a spectrum of environmental scares that have entered the public conscience during the decade. However, the problems of the Bight have included a number of dramatic incidents that have taken on the dimensions of environmental crises. Some of these perceived crises were not real but did illustrate the extent to which dramatic outbursts of public concern can lead to widespread misunderstandings. Nevertheless, real crises did in fact take place, ranging from chronic beach pollution incidents to the major oxygen depletion and beach closure of 1976. Were these crises of the 1970s more numerous and more serious than had occurred in prior decades or was it that situations were much the same and we simply looked and saw more? This is a difficult question to answer in view of the poor documentation prior to 1970. However, our best estimate must be that there are elements of truth in both views, that environmental crises were more frequent and severe during the seventies, but that the perception that these only began to occur during the seventies is also quite wrong.

During the seventies we have achieved much toward identifying and understanding the causes and effects of marine pollution in the New York Bight. In addition, much progress has been made in establishing the legislative and regulatory framework within which we can manage these causes and effects and prevent further degradation. One might even expect that the rates of pollutant discharge to the estuary and Bight may have decreased somewhat in response to implementation of the New York and New Jersey pollutant discharge elimination systems. However, we are not aware of any empirical evidence regarding trends in total pollutant loadings during the 1970s.

There is equally inadequate evidence for 10-year trends in water quality of the Bight. While an earlier study seemed to detect a slight downward trend in summer concentrations of dissolved oxygen in bottom waters of the apex from 1949 through 1974 (O'Connor, D.J., et al., 1977), Swanson et al. (1979) found that dissolved oxygen concentrations in recent years did not indicate a trend. Existing information on toxicants in Bight waters or sediments is inadequate to assess trends during the 1970s. Trends in biotic effects would be even more difficult to detect, and no such trends are evident. Somewhat more extensive measurements of bacterial pathogen indicators do not illustrate marked trends in their concentration or distribution.

We have now reached a point where many hard choices have to be made concerning the next steps to be taken in managing the New York Bight ecosystem with respect to pollution. The New York-New Jersey metropolitan area is a region of extreme stress in terms of both economic resources and intensity of land and water use. Therefore, mistakes in simply shifting pollutant loads from the ocean to land or air can have particularly high monetary and environmental costs. While some adjustments in the disposal media of existing wastes may be appropriate, it seems probable that the most useful improvements in waste management strategies must involve toxicant control and recycling or burning for energy use before the toxicants are greatly diluted in water. Given the increasing value of reclaimed energy and toxic materials and inexpensive reclamation technologies now available, all waste generating facilities, including households, should be induced to practice such source control.

While environmental science can be expected to provide openly some of the information required for waste management decisions, reliable information on more readily measured features of waste management is not always available. For instance, the quantities of pollutants liberated by specific industries or regions are often estimated incompletely and roughly; and reliable costs for, and effectiveness measures of, treatment processes are often difficult to gain. Wise decisions regarding waste management depend upon reliable knowledge of treatment effectiveness and costs just as importantly as upon knowledge of fate and effects in the Bight. Regulatory agencies should have adequate authority and resources to compile and maintain this information on a regular basis. The U.S. Coast Guard's Pollution Incident Reporting System is an illustration of such a reliable data base for oil and hazardous wastes spilled from vessels.

Although continuing insight can be expected in our knowledge of pollutant impacts, unrealistic expectations must not be used as the basis for deferring management actions. For instance, if an ecological or public health impact is judged unacceptable, it is seldom useful in the Bight environment to ask which pollutant or which pollutant source is "responsible." The New York Bight receives polluting wastes from an unusually large number of sources, and they are effectively mixed in coastal waters, sediments, and biota. Essentially all pollutants from all pollutant sources contribute to the observed biotic effects in the Bight; all pollutant sources are therefore responsible for ecosystem degradation. Their relative importance is a function of the quantity, toxicity, and biological availability of their pollutant composition. Thus, apart from exceptional situations, it will not be possible to identify a single chemical or pollutant source as responsible for an observed effect. If ecological effects in the Bight are viewed as socially unacceptable, effective remedial action can be taken without unrealistic attempts to first fix blame upon any single chemical or type of waste.

REFERENCES

Anderson, J.W. 1981. The transport of petroleum hydrocarbons from sediments to benthos and the potential effects. In: Ecological stress and the New York Bight: science and management, G.F. Mayer, ed. The Estuarine Research Federation, Columbia, SC. In press.

Beardsley, R.C., W.C. Boicourt, and D.V. Hansen. 1976. Physical oceanography of the Middle Atlantic Bight. In: Middle Atlantic Continental Shelf and New York Bight, M.G. Gross, ed. Amer. Soc. Limnol. Oceanogr. Spec. Symp. 2. Ann Arbor, MI. pp. 20–34.

Bird, D. 1973. Divers confirm finding of sludge a half mile off L.I. New York Times, December 29.

Blumm, M.C. 1980. Chapter 9. Implementing the Clean Water Act: progress, problems and possibilities. In: Water quality administration, B. Lamb, ed. Ann Arbor Science, Ann Arbor, MI.

Boddewyn, J.J. 1981. Economic development and the Port of New York. NOAA Special Report Series. NOAA, Rockville, MD. In press.

Buelow, R.W., B.H. Pringle, and J.L. Verber. 1968. A preliminary investigation of waste disposal in the New York Bight. Department of Health, Education and Welfare, Public Health Service, Northeast Marine Health Science Laboratory. Mimeo.

Cabelli, V.J. 1981. Bacterial and viral pathogens. In: Public Health MESA New York Bight Atlas Monograph 32. New York Sea Grant Institute, Albany, NY. In press.

Carls, E.G. 1978. Recreation. MESA New York Bight Atlas Monograph 19. New York Sea Grant Institute. Albany, NY.

Carroll, R. 1974. Experts in dispute over when sludge will hit beaches. Daily News, March 23:7.

CEQ (Council on Environmental Quality). 1970. Ocean dumping, a national policy. Council on Environmental Quality, Washington, DC.

Drake, D. 1974. Suspended particulate matter in the New York Bight apex: September–November 1973. NOAA-TR-ERL 318 MESA-1, Washington, DC.

Duce, R.A., G.T. Wallace, and B.J. Ray. 1976. Atmospheric trace metals over the New York Bight. NOAA Technical Report ERL 361-MESA 4. NOAA, Environmental Research Laboratories, Boulder, CO.

Energy and Environmental Analysis, Inc. 1975. Assessment of the National Pollutant Discharge Elimination System of Public Law 92-500. EEA Inc., Boston, MA.

Farrington, J.W., and B.W. Tripp. 1979. Hydrocarbons in western north Atlantic surface sediments. Geochim. Cosmochim. Acta 41:1627–1641.

Freeland, G.L., and D.J.P. Swift. 1978. Surficial sediments. MESA New York Bight Atlas Monograph 10. New York Sea Grant Institute, Albany, NY.

Gibson, C.I., L.L. Ciaccio, B. Ketchum, and A.D. Michael. 1979. Petroleum hydrocarbons subpanel report. In: Chemical pollutants of the New York Bight: priorities for research, J.S. O'Connor and H.M. Stanford, eds. NOAA, MESA, Environmental Research Laboratories, Boulder, CO.

Goldberg, E.D., ed. 1979. Assimilative capacity of U.S. coastal waters for pollutants. Proceedings of a workshop, Crystal Mountain, WA. NOAA, Environmental Research Laboratories, Boulder, CO.

Gordon, R.B., W.F. Bohlen, H.J. Bokuniewicz, M. de Picciotto, J. Johnson, K. Kamlet, T.F. McKinney, J.R. Schubel, D. Suszkowski, and T. Wright. 1981. Management of dredged material. In: Ecological stress and the New York Bight: science and management, G.F. Mayer, ed. The Estuarine Research Federation, Columbia, SC. In press.

Gross, M.G. 1976. Waste disposal. MESA New York Bight Atlas Monograph 26. New York Sea Grant Institute, Albany, NY.

Grosslein, M.D., and T. Azarovitz. 1981. Fish distribution. MESA New York Bight Atlas Monograph 15. New York Sea Grant Institute. Albany, NY. In press.

Gunnerson, C.G. 1981. The New York Bight ecosystem. In: Marine Environmental Pollution, 2. Dumping and mining, R.A. Geyer, ed. Elsevier Scientific Publishing Co., Amsterdam. In press.

Kamlet, K.S. 1981. Dredge spoil ocean dumping. Perspectives on legal and environmental impacts. In: Proceedings of the 2d International Ocean Dumping Symposium, Woods Hole, MA. April 15–18, 1980. In press.

Kidder, T. 1975. Sludge. Atlantic Monthly 235:62–68, 70.

Kline, P. 1974. Is dumping garbage in the ocean a menace? Daily News, February 17.

Kneip, T., N. Cutshall, R. Field, F.C. Hart, P.J. Lioy, J. Mancini, J. Mueller, C. Sobotowski, and J. Szeligowski. 1981. Non-point sources. In: Ecological stress and the New York Bight: science and management, G.F. Mayer, ed. The Estuarine Research Federation, Columbia, SC. In press.

Loop, A.S. 1964. History and development of sewage treatment in New York City. Department of Health, City of New York, New York, NY.

MacLeod, W.D., Jr., L.S. Ramos, A.J. Friedman, D.G. Burrows, P.G. Prohaska, D.L. Fisher, and D.W. Brown. In press. Analysis of residual chlorinated hydrocarbons, aromatic hydrocarbons and related compounds in selected sources, sinks and biota of New York Bight. NOAA, Office of Marine Pollution Assessment, Technical Memorandum.

Madden, R.L. 1970. 29 in House urge port dumping ban. New York Times, February 12:1,42.

Malone, T.C., W. Esaias, and P. Falkowski. 1979. Plankton dynamics and nutrient cycling. In: Oxygen depletion and associated benthic mortalities in New York Bight, 1976, R.L. Swanson and C.J. Sindermann, eds. NOAA Professional Paper 11. Rockville, MD. pp. 193–229.

Mayer, D.A., D.V. Hansen, and D.A. Ortman. 1979. Long-term current and temperature observations on the Middle Atlantic shelf. J. Geophys. Res. 84(C4):1776–1792.

MESA. 1975. Ocean dumping in the New York Bight. NOAA Technical Report, ERL 321–MESA 2. NOAA, Environmental Research Laboratories, Boulder, CO.

MESA. 1977. Long Island Beach Pollution: June 1976. MESA Special Report. NOAA, Environmental Research Laboratories, Boulder, CO.

Mueller, J.A., J.S. Jeris, A.R. Anderson, and C.F. Hughes. 1976. Contaminant inputs to the New York Bight. NOAA Technical Memorandum ERL MESA-6. NOAA, Environmental Research Laboratories, Boulder, CO.

NAS (National Academy of Sciences). 1975. Petroleum in the marine environment. Workshop on Inputs, Fates and the Effects of Petroleum in the Marine Environment. National Academy of Sciences, Washington, DC.

New York City. 1979. Section 208 Areawide Waste Treatment Management Planning Program. New York City Department of Environmental Protection, New York.

O'Connor, D.J., R.V. Thomann, and H.J. Salas. 1977. Water quality. MESA New York Bight Atlas Monograph 27. New York Sea Grant Institute. Albany, NY.

O'Connor, J.M., J.B. Klotz, and T.J. Kneip. 1981. Sources, sinks and distribution of organic contaminants in the New York Bight Ecosystem. In: Ecological stress and the New York Bight: science and management, G.F. Mayer, ed. The Estuarine Research Federation, Columbia, SC. In press.

O'Connor, J.M., T.J. Kneip, R.A. Greig, F.P. Thurberg, J.S. O'Connor, H.M. Stanford, and L.S. Ramos. 1980. Organic contaminants in the New York Bight ecosystem. Mar. Poll. Bull. In press.

O'Connor, J.S. 1976. Contaminant effects on biota of the New York Bight. Proceedings of the Annual Session Gulf and Caribbean Fish. Institute. pp. 50–63.

O'Connor, J.S. 1981. Resources and ecosystem well-being in the New York Bight. NOAA Special Report Series. NOAA, Rockville, MD. In press.

O'Connor, J.S., and H.M. Stanford, eds. 1979. Chemical pollutants of the New York Bight: priorities for research. NOAA, MESA, Environmental Research Laboratories, Boulder, CO.

Pearson, H. 1974. Mystery sludge: the plot thickens. Newsday, January 19.

Pearson, H. 1974. Sea of sludge gets one-year lease on life. Newsday, February 12.

Pearson, H. 1974. Sludge half mile off L. Beach. Newsday, July 11.

Segar, D.A., and G.A. Berberian. 1976. Oxygen depletion in the New York Bight apex: causes and consequences. In: Middle Atlantic Continental Shelf and New York Bight, M.G. Gross, ed. Amer. Soc. Limnol. Oceanogr. Spec. Symp. 2. Ann Arbor, MI.

Segar, D.A., and A.Y. Cantillo. 1975. Trace metals in the New York Bight. Abstracts from the Special Symposium on the Middle Atlantic Continental Shelf and New York Bight, November 3–5, 1975, Amer. Soc. Limnol. Oceanogr. Ann Arbor, MI. p. 62.

Segar, D.A., G.A. Berberian, and P.G. Hatcher. 1975. Oxygen depletion in the New York Bight apex: causes and consequences. Abstracts from the Special Symposium on the Middle Atlantic Continental Shelf and New York Bight, November 3–5, 1975, Amer. Soc. Limnol. Oceanogr. Ann Arbor, MI. p. 61.

Sharov, L. 1973. A chain of reactions on sludge discovery. Newsday, December 12.

Souci, G. 1974. Here come de sludge. Audubon 76(4):108–113.

Squires, D.F. 1981. The Bight of the Big Apple. NOAA Special Report Series. NOAA, Rockville, MD. In press.

Stanford, H.M., J.S. O'Connor, and R.L. Swanson. 1981. The effects of ocean dumping on the New York Bight ecosystem. In: Ocean dumping of industrial wastes, B.H. Ketchum, D.R. Kester, and P.K. Park, eds. Plenum Press, NY. In press.

Suszkowski, D.J. 1973. Sewage pollution in New York Harbor: a historical perspective. M.S. Thesis, State University of New York, Stony Brook, NY.

Swanson, R.L. 1977. Status of ocean dumping research in New York Bight. J. Waterway, Port, Coastal and Ocean Div., ASCE, 103(WW1):9–24.

Swanson, R.L., and C.J. Sindermann, eds. 1979. Oxygen depletion and associated benthic mortalities in New York Bight, 1976. NOAA Professional Paper 11. NOAA, Rockville, MD.

Swanson, R.L., C.J. Sindermann, and G. Han. 1979. Oxygen depletion and the future: an evaluation. In: Oxygen depletion and associated benthic mortalities in New York Bight, 1976, R.L. Swanson and C.J. Sindermann, eds. NOAA Professional Paper 11. NOAA, Rockville, MD.

Tietjen, J. 1980. Population structure and species composition of the free-living nematodes inhabiting sand of the New York Bight apex. Estuar. Coastal Mar. Sci. 10:61–73.

U.S. Environmental Protection Agency. 1974. Briefing report: ocean dumping in the New York Bight since 1973. U.S. Environmental Protection Agency, Region II, New York, NY.

U.S. Environmental Protection Agency. 1976. Environmental impact statement on the ocean dumping of sewage sludge in the New York Bight. Draft, February 1976. EPA-2-NY, NJ - Ocean Dumping 76. U.S. Environmental Protection Agency, Region II, New York, NY.

U.S. Environmental Protection Agency. 1978. Environmental impact statement on the ocean dumping of sewage sludge in the New York Bight. Final, September 1978, U.S. Environmental Protection Agency, Region II, New York, NY.

U.S. Environmental Protection Agency/Corps of Engineers. 1977. Ecological evaluation of proposed discharge of dredged material into ocean waters: implementation manual for Section 103 of Public Law 92-532 (Marine Protection Research and Sanctuaries Act of 1972). U.S. Environmental Protection Agency/Corps of Engineers Technical Committee on Criteria for Dredged and Fill Material. U.S. Army Engineer Waterways Experimental Station, Vicksburg, MS.

Verber, J.L. 1976. Safe shellfish from the sea. In: Middle Atlantic Continental Shelf and New York Bight, M.G. Gross, ed. Amer. Soc. Limnol. Oceanogr., Spec. Sympos. 2. Ann Arbor, MI. pp. 434–441.

Walsh, J.J. 1980. Shelf-sea ecosystems. In: Analysis of marine ecosystems, A.R. Longhurst, ed. Academic Press, NY. In press.

Wigley, R.L., and R.B. Theroux. 1976. Macroinvertebrate fauna of the Middle Atlantic Bight Region. Part II. Faunal composition and quantitative distribution. NOAA, Northeast Fisheries Center, Woods Hole, MA.

MAN'S IMPACT ON THE COASTAL ENVIRONMENT: NUTRIENTS IN THE MARINE ENVIRONMENT

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INTRODUCTION

The waste of human metabolism is rich in organic matter and in the fertilizing nutrients essential for plant growth. Civilized people dilute this potentially valuable natural resource with excessive volumes of carefully purified drinking water, add a witch's brew of toxic chemicals, collect the resultant mixture from large areas of densely populated land to concentrate it in one small location for disposal, and then complain bitterly that it is expensive, if not impossible, to discharge the product into the environment without damage to the ecosystem. For over a century the hazards posed to public health by sewage have been recognized, and it is sterilized to prevent the spread of waterborne disease. It was also early recognized that this mixture contained large amounts of organic matter and that excessive discharge into an isolated part of the aquatic environment would remove all available oxygen from the water and lead to putrefaction and the production of hydrogen sulfide with its obnoxious odor. Treatment plants were built to solve these two problems. The sewage was separated into a relatively clear though not drinkable effluent and a watery quasi-solid containing much of the particulate organic material and a major share of the toxic materials that were added in the collection system. In this way, two disposal problems were created in place of one.

In retrospect, it appears that little has been accomplished that is admirable or desirable, though it is obvious that modern cities could not exist without their sewers. Today it is known that the apparently clear and innocuous effluent contains most of the elements essential for plant growth that were in the original mixture. This effluent is a biostimulant or fertilizer, and when this is added to the aquatic environment, the natural plant populations quickly produce about the same amount of organic material as that which was removed at great expense in the treatment plant. The semisolid sludge, which contains most of the organic material present in the original mixture, could be used as a soil conditioner and fertilizer except that the excessive toxics it contains make this disposal option impractical.

Thus the human population finds itself in a dilemma. Efforts to solve one set of problems have created other problems that are at least as serious. These problems are not confined to any one country; they are global in extent and are found wherever people gather together into densely populated urban areas. If the human population were spread uniformly over the entire habitable area of earth, there would be little problem. For large areas, the "night-soil" would increase agricultural production, albeit with the hazard of disease propagation. For more fastidious people, onsite disposal is practical so long as each family unit is far enough removed from its neighbors. Obviously, demands of modern civilization make this option impractical.

The world oceans have long been considered a valuable resource for the disposal of human waste materials based on the philosophy that "the solution to pollution is dilution." As Goldberg points out in this volume, it is critical to assess the assimilative capacity of the marine environment in order to avoid making foolish mistakes. To provide perspective on this problem, it can be estimated that it would take about twenty thousand years to double the total available nitrogen content of seawater if all of the sewage from today's human population were discharged into the ocean. This assumes complete worldwide mixing and that there would obviously be local concentrations greatly in excess of the average where the material is discharged. It assumes, furthermore, that there would be no biological adjustment of the system, such as by the sequestering of some of the added materials to the sediment or by denitrification. And, of course, the problem would be exacerbated by further growth of the world population.

The fact that there is a large capacity does not answer the fundamental question. How much of this capacity can be safely used? Would doubling the available nitrogen content of the sea be acceptable? Would 10 percent? 1 percent? Would the added nutrients in sewage stimulate marine productivity so that the harvest from the sea could be increased? Could enough material be harvested to keep the system in a continuous steady-state balance?[*] How can local deleterious concentrations that upset the natural ecosystem and do more damage than they do good be avoided?

THE NATURAL NUTRIENT CYCLE

Nitrogen and phosphorus are the elements that most frequently limit plant growth in aquatic ecosystems. For diatoms, silica, which is essential for the formation of the shell, is also sometimes limiting. Various trace elements such as boron, cobalt, copper, iron, molybdenum, and zinc are also required for plant growth but are rarely limiting in seawater. Other elements such as carbon, potassium, and sulfur are essential components of living material, and they are present in excess quantities for plant requirements in the sea. The scope of this paper will focus upon nitrogen and phosphorus as the most critical elements.[†]

Normal Proportion of Elements

In the production of organic matter in photosynthesis and the subsequent decomposition of this material, the major elements are utilized or released in statistically similar proportions to one another (Redfield et al., 1963). In seawater, these proportions are:

$$-\mathrm{O{:}C{:}N{:}P} = -276{:}106{:}15{:}1 \text{ (by atoms)}$$

or

$$-142{:}41{:}6.8{:}1 \text{ (by weight)}$$

The oxygen value is negative in the above relations because it is released in photosynthesis when the other elements are being absorbed, and it is utilized in decomposition when the other elements are being released. Except for carbon, which is present in seawater in considerable excess of the needs of the plankton, these proportions are remarkably close to the concentrations available in seawater. The data in Table 1 (Redfield et al., 1963) show a comparison between availability and utilization. Except for carbon, which is available at nearly 10 times the amount required by the phytoplankton, the other elements are present in average seawater in the approximate proportions they are needed for the ecological cycle.

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[*] The present annual fish harvest of about 60 million metric tons removes about 1.3×10^9 kg N/yr^{-1} assuming 120 kg protein containing 17.8 percent N per ton. This is about 3 percent of the nitrogen in the human sewage of the present world population. The importance of marine protein to human nutrition is discussed by Ketchum (1973).

[†] Phytoplankton production is, of course, also dependent on other environmental characteristics, especially the transparency of the water, which determines the depth of the euphotic zone, the depth of the mixed layer, salinity, and temperature.

Table 1. Availability of Nutrient Elements and of Oxygen in "Average" Seawater (S=34.7‰; T=2°C) and the Ratios of Their Availability and Utilization by Plankton (after Redfield, et al., 1963)

Substance	Availability μg atoms 1^{-1}	Ratio	Utilization by plankton ratio	Availability vs. utilization, ratio
Phosphorus	2.3	1	1	1
Nitrogen	34.5	15	16	0.94
Carbon	2340	1017	106	9.6
Oxyben at saturation	735	320	-276	1.16

The oxygen content is of particular interest in terms of polluting the sea with added nutrients. Since photosynthesis takes place in the near surface layers, the oxygen produced is released to the atmosphere while the organic matter produced can sink to deeper layers, isolated from the surface, where it is decomposed. Table I shows a very small margin of safety in terms of preventing the complete removal of oxygen from the water and the production of anoxic and objectionable conditions.

The concentrations of nitrogen and phosphorus shown in Table I are characteristic of the average deep water of the world oceans and are considerably higher than the concentrations that would be found in surface layers of coastal or estuarine waters. The oxygen content of the warm, shallow waters would also be less because oxygen solubility decreases with an increase in temperature. Nevertheless, the numbers are useful to set limits that should not be exceeded for nitrogen and phosphorus in order to avoid the development of anoxic conditions.

The Limiting Nutrient

These ratios do not represent the proportions in which the elements are available in coastal or estuarine seawater but rather the ratios of change in their concentrations that result from biological activity. The phytoplankton biomass will be limited by the nutrient available in the environment in the smallest quantity relative to the requirement of the plants, provided all other factors such as light and temperature are favorable. It will be recognized that the element may not be limiting at the time of observation if all of the essential elements are present in excess. The limiting element will be the first nutrient to become exhausted following growth of the phytoplankton population.

In freshwater, phosphorus is commonly the limiting nutrient, and Miller et al. (1975) have suggested that when freshwater contains a N:P ratio greater than 11.3:1 by weight (25:1 by atoms) the water may be considered phosphorus-limited while water containing lower N:P ratios can be considered nitrogen-limited for algal growth. In freshwater, consequently, removal of excess phosphorus from sewage wastes has resulted in considerable improvement in the quality of the receiving water.

In contrast to this, the common limiting nutrient in marine waters is nitrogen (Ryther and Dunstan, 1971; Eppley et al., 1971; Thomas et al., 1974; Doig and Martin, 1974; Goldman, 1976). In marine waters a N:P atomic ratio lower than 15:1 (6.8:1 by weight) implies nitrogen limitation of the phytoplankton production.

In New England coastal waters, Ketchum et al. (1958) showed that the ratio of *change* of nitrogen and phosphorus in the near surface waters was 15:1 by atoms but that the waters contained a residual phosphorus content of 0.32 to 0.55 μg - at l^{-1} when the nitrogen was virtually exhausted. This excess phosphorus is reflected in low N:P ratios of concentration. The ratio of inorganic compounds of nitrogen and phosphorus (N:P) in water samples collected at 10 m (33 ft) at several locations and various times of year ranged from 1.21 to 7.08 (by atoms). In contrast, the total N:P ratio for the same samples ranged from 14.9 to 30.2. Thus the organic matter is greatly enriched in nitrogen when compared with the inorganic source of nitrogen and phosphorus in the water. In the decomposition of organic matter, phosphorus is released more rapidly than is nitrogen, contributing to the low N:P ratio of concentration in the water.

Goldman (1976) has studied the growth and composition of a marine diatom, *Phaeodactylum tricornutum*, in seawater enriched with sewage wastes or with added nutrients. Some of his results are illustrated in Figure 1. The typical N:P ratio in the wastewater ranged from 5 to 10 (by atoms), whereas the ratio in the algae ranged from 10 to 20. All of the cultures in Goldman's experiments were nitrogen-limited as evidenced by the fact that the N:P ratio in the algae was consistently greater than the N:P ratio in the medium in which they were grown.

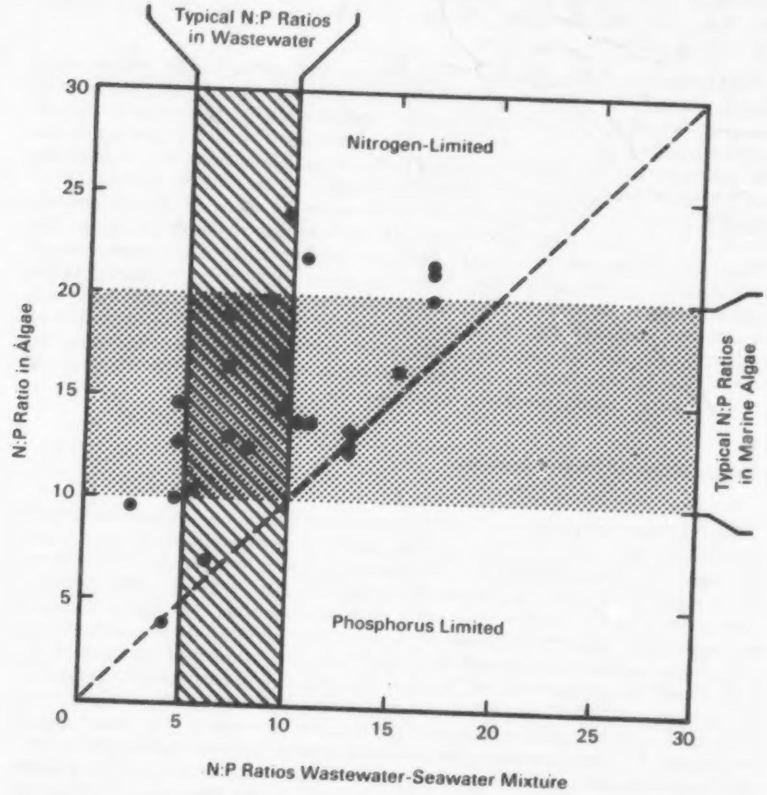

Figure 1. Comparison of atomic N:P ratios observed in wastewater-seawater mixtures and in *Phaeodactylum tricornutum* (from Goldman, 1976).

The U.S. Environmental Protection Agency (1974) has developed a marine algal assay procedure designed to evaluate the effects of nutrients on the growth of marine phytoplankton. This is a batch test in which spikes of a suspected limiting nutrient are added to the pure algal culture and the total biomass produced after a fixed period of time, generally 2 weeks, is determined as a function of the added nutrient. The total biomass produced by cultures of *Dunaliella tertiolecta* in an artificial seawater medium was shown to be directly proportional to the added concentrations of phosphorus or of nitrogen when the basic medium was prepared without these elements. The tests were performed at various salinities, and it was found that there was no significant effect of salinity in the range of 16 to 35‰.

The assay test was also applied to estuarine waters at high and low tides with interesting results. Phosphorus was consistently the limiting nutrient when the samples were taken from the estuary at low tide, whereas nitrogen was limiting in samples collected at high tide. The salinity showed the expected changes with the stage of the tide, but the relationship of the production of the organic matter with salinity was not significant. The nitrogen content of the water samples at low tide was consistently higher than that at high tide, whereas the phosphorus content showed little change. This confirms the generalization that phosphorus is commonly the limiting nutrient in the freshwater that predominates in these estuaries in the low tide samples, whereas nitrogen is commonly limiting in the marine waters that dominated at the time of high tide.

A change in the concentration of a limiting nutrient could affect the production of organic matter by phytoplankton either by changing the rate of growth or by changing the total biomass that can be supported. Goldman (1979) states that marine phytoplankton have such a high affinity for nutrients such as nitrogen and phosphorus that these chemicals are frequently below detectable levels over virtually the entire growth rate spectrum. It is impossible to evaluate the effect of an essential element on the rate of growth in a batch test because the algae are continuously exposed to a changing concentration as they assimilate the element and grow. The effect of a limiting nutrient on the growth rate can be evaluated in a chemostat culture in which a continuous supply of fresh medium is provided. The growth rate in such a culture is proportional to the rate of dilution at a steady-state cell concentration. It seems unlikely that the rate of growth of phytoplankton in the natural environment, particularly in estuaries and coastal waters, is limited by the availability of nutrients, but the biomass that can be produced is certainly proportional to the amount of the limiting nutrient available for the production.

Composition of Phytoplankton

The fact that the normal Redfield N:P atomic ratio of 15:1 is consistently found in deep ocean waters and in the ratio of change of these elements in surface waters does not imply that the composition of phytoplankton cells is immutable. Under laboratory conditions, the N:P ratio of the cell can be varied as a function of the composition of the culture medium and other growth conditions. The observations of Ketchum (1939) that cells deficient in phosphorus are produced when grown under phosphorus limitations of growth have been repeatedly confirmed (Goldman et al., 1979; Perry, 1976; Fuhs et al., 1972). Goldman et al. (1979) conclude that the typical Redfield ratio of C:N:P equals 106:15:1 is approached in culture only at high growth rates relative to the potential maximum growth rate. This implies that, under natural conditions, the phytoplankton that show this typical ratio are growing at or near their maximum rates, even though the environmental concentrations of nitrogen and phosphorus are low as is the phytoplankton biomass at any given time. They discuss the dynamic conditions that make a maximum growth rate possible under these conditions.

DOMESTIC (HUMAN) WASTES

The problems of the disposal of domestic pollution are global in extent, but the impact on the environment is localized because of collection and release to a limited area. Segar (in press) has compiled data for the release of contaminants in various populated parts of the world. The estimated amounts of nitrogen and phosphorus are shown in Table 2. The nitrogen contamination of these marine areas ranges from 8,000 to 33,000 tons per million population per year, and the range of phosphorus contamination is from 900 to 3,600. Both of these vary by a factor of four but not in parallel with each other; the Baltic Sea appears to receive an excessive amount of nitrogen. Excluding the Baltic and the Seto Inland Sea, the ratio of nitrogen to phosphorus by weight ranges from 2.78 to 6.54 (6.14 to 14.45 by atoms). The North Sea, Seto Inland Sea, and the Baltic have an excess of nitrogen or an adequate amount to meet the relative requirements of the phytoplankton. The New York Bight, the Mediterranean, and the Irish Sea appear to be relatively deficient in nitrogen. The Baltic Sea is the freshest of these coastal areas, and, like most fresh waters, it seems to be deficient in phosphorus, relative to nitrogen, in terms of the needs of the phytoplankton populations.

Table 2. Contaminant Inputs and N:P Ratios for Selected Ocean Regions (after Segar, in press)

	Input tons/yr/10^6 people			
Region	Nitrogen	Phosphorus	N:P by wt.	N:P by atoms
New York Bight	7,600	2,000	3.8	8.4
Mediterranean	10,000	3,600	2.78	6.14
Seto Inland Sea	8,000	900	8.89	19.64
Baltic	33,000	950	34.7	76.7
North Sea	17,000	2,600	6.54	14.45
Irish Sea	10,000	2,800	3.57	7.89

Traditional sewage treatment was developed to achieve two major objectives: the prevention of waterborne diseases and the removal of organic matter that contributes to the biological oxygen demand (BOD) and could thus lead to anoxic conditions in the environment. These conventional treatment methods were not designed to remove the essential plant nutrients from the effluent and they have, indeed, little effect on them. The average concentrations of nitrogen and phosphorus in the effluent from a number of treatment plants is shown in Table 3A (Mancini et al., in press) and the typical wastewater concentrations of these elements in comparison to the BOD of the wastewater is shown in Table 3B. In comparison to the requirements of the phytoplankton in the marine environment, all of these samples are clearly very deficient in nitrogen relative to phosphorus. The ratio N:P by weight for all of these samples ranges from 2.21 to 6.67 (4.90 to 14.73 by atoms).

The low N:P ratio in wastewater can be considered, of course, either as a deficiency of nitrogen or as an excessive amount of phosphorus. It has been estimated that human wastes account for 30 to 50 percent of the phosphorus in modern domestic wastewater. The balance is accounted for by the use of phosphorus-enriched detergents (Mancini et al., in press). Mancini et al. present the data that are summarized in Table 4 showing the effects of phosphorus detergent bans or phosphorus removal on the phosphorus content of wastewater and on the N:P ratios of the effluents. The banning of detergents or removal of phosphorus in the treatment process increases the N:P ratio in the effluent, but the effluent is still nitrogen-deficient relative to the requirements of marine phytoplankton.

Table 3A. Total Nitrogen and Phosphorus Concentration (Median) in Wastewater Effluents Following Four Conventional Treatment Processes (after Mancini et al., in press)

	Treatment type			
	Primary	Trickling filter	Activated sludge	Stabilization pond
No. of plants sampled	55	244	244	149
Total P (mg 1^{-1})	6.6 ± 0.66	6.9 ± 0.28	5.8 ± 0.29	5.2 ± 0.45
Total N (mg 1^{-1})	22.4 ± 1.30	16.4 ± 0.54	13.6±0.62	11.5 ± 0.84
N:P (weight)	3.39	2.38	2.34	2.21
N:P (atoms)	7.52	5.26	5.19	4.90

Table 3B. Typical Wastewater Characteristics (after Mueller, et al., 1976)

	Concentration (mg 1^{-1}) for		
	New York City raw sewage	New Jersey primary effluent	New York City secondary effluent
BOD_5	131	158	36
Total P	4.70	6.14	3.30
Total N	21.7	22*	22*
N:P (weight)	4.62	3.58	6.67
N:P (atoms)	10.20	7.92	14.73

***Average of primary and secondary effluent concentrations.**

As far as the marine environment is concerned, control of the phosphorus content of the effluent cannot be expected to limit effectively the excessive plant growths in the environment into which the effluent is discharged. Since nitrogen is commonly the most critical element limiting phytoplankton production in the marine environment, removal of nitrogen from the wastewater effluents would be expected to be the most effective way to prevent excessive phytoplankton growth. Various methods are available for nitrogen removal from sewage effluents. Their effectiveness is summarized in Table 5 (Mancini et al., in press). Conventional primary and secondary treatments remove some of the organic nitrogen from the sewage but have little or no effect on inorganic nitrogen. Various advanced wastewater treatment processes can remove as much as 90 percent of the total nitrogen. Even such comparatively simple treatments as oxidation ponds can be very effective in removing nitrogen, provided the organic material produced in the photosynthetic process in these ponds is removed as particulate matter before the effluent is released to the environment.

If the effluent is released to the environment without control of the elements that stimulate plant growth, the natural photosynthetic process in the environment will recreate the organic material that was removed or decomposed at considerable expense in the treatment process. There is a time lag in achieving this in the environment the duration of which is dependent upon all of the environmental factors that

Table 4. Effect of P-Detergent Ban or P Removal on the N:P Ratio of Wastewaters (after Mancini, et al., in press)

Conditions	No. plants sampled	Total P mg 1^{-1}	Total N mg 1^{-1}	N:P by wt.	N:P by atoms
No P removal or detergent ban	709	7.0 ± 0.31	17.5 ± 0.9	2.5	5.53
P-detergent ban	25	4.6 ± 1.41	12.5 ± 2.78	2.72	6.01
P-removal	33	2.7 ± 0.61	13.7 ± 2.57	5.07	11.21

control phytoplankton growth. These factors include not only nutrient content but also the rate of the circulation; the rate of grazing by herbivorous animals; temperature and the transparency of the water, which determines the depth of the euphotic zone; and the total amount of photosynthesis that can take place in the water column.

CIRCULATION AND MIXING

Circulation of the water in the environment in which the sewage is released will determine the rate of dispersion and dilution of the effluent, and the advective processes will transport the contaminated water away from the point of discharge. The rates of these processes depend upon the geomorphology of the estuary, the characteristics of the tidal regime, and the river flow. These are unique characteristics of each estuary and must be separately evaluated for each. Circulation and mixing within the estuary determine the fate of a pollutant, but these processes are commonly inadequately evaluated or poorly understood in the planning of marine disposal operations.

A conceptual or mathematical model of a specific estuary is desirable in order to understand clearly the dynamics of the circulation and to facilitate comparison with other estuaries. The early conceptual models were based upon the assumption of the steady-state distribution of properties (Tully, 1949; Ketchum, 1951a). The development of computer technology has made it possible to model transient conditions in the estuary. Some of the conceptual models are designed to take account of biological processes, and several of these are discussed by O'Connor (in press).

While each estuary is unique, certain generalizations apply to all estuaries in which the water supplied by river flow is diluted by seawater. There is a gradient of salinity from the virtually fresh water of the river to the salinity of coastal seawater, generally 30 to 32‰. The surface layers are fresher than the deeper waters within the estuary, and these two layers are separated by a density discontinuity that generally depends on the salinity distribution (the halocline). A density-driven circulation pattern is established within the estuary so that the net transport during a complete tidal cycle in the surface, fresher layer is seaward and the net transport in the deeper layer is landward. Under steady-state conditions, the net effect of these two flows is to transport seaward a volume of freshwater equal to the volume introduced by the river during a complete tidal cycle. The net transport of salt through any complete cross-section of the estuary is equal to zero. When the river flow is not constant, the steady-state conditions will not be met. When the rate of river flow is declining, salinity will increase at any location within the estuary and there will be an excess inflow of seawater. When the river flow is increasing, the estuary will become fresher and there will be a net transport of both salt water and freshwater seaward.

Any conservative soluble pollutant that is added to the estuary will be distributed, diluted, and transported by the same processes that control the distribution of freshwater and salt water. If the pollutant is introduced at a mid-point in the estuary and

Table 5. Removal of Nitrogen from Sewage Effluents (after Mancini, et al., in press)

Treatment process	Effect on constituent			Removal of total Nitrogen Entering Process (Percent)
	Organic-N	Ammonia-N	Nitrate-N	
Conventional treatment processes				
Primary	10-20% removed	no effect	no effect	5-10
Secondary	15-20% removed urea→NH_3/NH_4^+	<10% removed	nil	10-20
Advanced wastewater treatment processes				
Filtration	30-95% removed	nil	nil	20-40
Carbon sorption	30-50% removed	nil	nil	10-20
Electrodialysis	100% of suspended organic N removed	40% removed	40% removed	35-45
Reverse osmosis	100% of suspended organic N removed	85% removed	85% removed	80-90
Chemical coagulation	50-70% removed	nil	nil	20-30
Other nitrogen removal processes				
Selective ion exchange for nitrate	nil	nil	75-90% removed	70-90
Oxidation ponds	partial transformation to NH_3/NH_4^+	partial removal by stripping	partial removal by nitrification-denitrification	20-90
Algae stripping	partial transformation to NH_3/NH_4^+	→cells	→cells	50-85
Bacterial assimilation	no effect	40-70% removed	limited effect	30-70

becomes quickly and uniformly mixed throughout the water column, the upstream distribution will be proportional to the upstream distribution of salt water and the downstream distribution will be proportional to the downstream distribution of freshwater (Ketchum et al., 1952; Ketchum, 1955; Ketchum, 1969).

One important consequence of these relationships is that the volume of water available for the dilution of a pollutant is considerably greater in the estuary than it would be in a river because of the participation of the salt water in the circulation. If, for example, the mixed water moving seaward is 50 percent fresh and 50 percent salt, it is clear that two volumes of the mixture must move seaward in order to remove the one volume of river flow that is the necessary net seaward transport across the cross-section. Also, as the salt water content of the mixture increases, the volume that must escape the system must also increase to carry one equivalent river flow seaward.

A numerical example using Pritchard's (1969) method of calculation may serve to clarify the process. It is presented in Table 6. This calculation is essentially a two-layered box model in which the estuary is divided into a number of segments along its length. The boundary between the upper and lower layers coincides with the boundary between the surface layer having a net nontidal flow directed seaward and the deeper layer having a net nontidal flow directed up-estuary. There is exchange between upper and lower layers by vertical eddy diffusion. The flux across any complete cross-section of the estuary in both the upper and lower layers is derived from the average salinity of these two layers assuming a steady-state salt distribution and providing for continuity of salt and volume. The calculated flux is relative to the flow of the river (R) in a given unit of time, commonly a complete tidal cycle. For the salinities arbitrarily chosen for Table 6, the seaward flux in the upper layer is three times the volume of river flow and the up-estuary flux in the lower layer is twice the river flow volume. The net flux is, as it must be in a steady-state distribution, equal to the volume introduced by the river in the time period selected.

In this example, both the upper layer and the lower layer are mixtures of freshwater and seawater, and the proportions of each, relative to offshore seawater, can also be calculated from the salinity (Ketchum, 1951b). This permits the calculation of the separate flux of salt water and freshwater in the system. There is a net flow of freshwater out of the segment that is equal to the flow of river water in the period of time selected. The flux of salt water seaward through the upper layer is exactly balanced by the flux of salt water landward in the deeper layer. These two conditions,

Table 6. Example of Calculation of the Horizontal Volume Flux, Relative to River Flow R, Through a Cross-Section of a Hypothetical Moderately Stratified Estuary

Condition	Upper Layer	Lower Layer	Net Flux
Salinity ‰ (S)	16	24	
Volume flux (Q)	3 x R (out)*	2 × R (in)†	1.0 R (out)
Saltwater fraction (F_s)§	0.5	0.75	
Freshwater fraction (F_f)§	0.5	0.25	
Saltwater flux	1.5 x R (out)	1.5 R (in)	0
Freshwater flux	1.5 x R (out)	0.5 R (in)	1.0 R (out)

*$Q_u = R (S_1/S_1 - S_u)$ and

†$Q_1 = R (S_u/S_1 - S_u)$ where Q is the flux, R is river flow, S is mean salinity of u the upper or 1 the lower layer (Pritchard, 1969).

§Relative to coastal, source seawater (σ) of 32‰ (Ketchum, 1951b); $F_s = Sx/\sigma$; $F_f = 1 - F_s = (\sigma - Sx/\sigma)$ where Sx is the mean salinity of the layer.

namely a net flux of freshwater through every complete cross-section equal to the amount of freshwater added at all upstream locations of the boundary selected and a net flux of salt water equal to zero, are the essential conditions for the maintenance of a steady-state distribution of salt and volume within the estuary.

When a pollutant is added to the estuary, it will be distributed by the same processes that distribute the salt water and freshwater. If the pollutant is introduced into the bottom layers, it will be carried in a net motion toward the head of the estuary in the lower layer flow. Turbulent mixing, which derives most of its energy from the ebb and flow of the tide, will disperse the pollutant horizontally in both a longitudinal and lateral direction, and it will be vertically dispersed into the surface layers where it is carried seaward again. Some of the pollutant in the surface layer is mixed downward into the deeper layer and will be carried again toward the head of the estuary. For a single discharge, there will be a peak concentration that will both gradually diminish and move seaward on each successive tidal cycle. If the pollutant is "conservative," that is, not decomposed, changed by biological activity, or sequestered in the bottom deposits, the changes in its distribution can be derived from the known salinity distribution of the estuary (O'Connor and Thomann, 1971).

In terms of domestic pollution, the continuous discharge at a more or less steady rate of the pollutant into the estuary is of great concern. With time, such a pollutant will become uniformly mixed laterally and, in proportion to the flux in each layer of a stratified estuary, will be transported landward in the deeper water and seaward in the surface layers. Upstream from the point of introduction, the concentration of the pollutant will be greater in the deep layers than in the surface layers, while seaward of the point of introduction, the converse will be true. The pollutant is ultimately flushed from the estuary in the seaward-directed flow of the surface layers.

Based upon these fundamental considerations, a comparison among estuaries, rivers, and lakes as places for the disposal of pollutants may be useful. In a river, the volume available for the dilution of a pollutant is equal to the volume of river flow in a unit period of time. In contrast, in an estuary the available diluting volume is augmented by the participation of seawater in the circulation, and the volume increases as the salinity in the sample increases. This occurs progressively as one moves from the location of the maximum penetration of salt into the estuary toward the mouth of the estuary. This has important implications in terms of the selection of a location for an outfall in an estuary. A downstream placement of the outfall location will always decrease the upstream concentration of the pollutant but will have no effect on the downstream distribution (Ketchum, 1955). Thus, if the water quality in the river or at the head of the estuary is of principal importance, the outfall should be placed as far downstream as is economically practicable to obtain the maximum improvement.

When the pollutant is biologically degradable or otherwise changed with time, the situation is somewhat more complicated because the residence time in various parts of the estuary must be considered (Ketchum, 1955; O'Connor and Thomann, 1971; O'Connor, in press). For such a time variable pollutant, the upstream concentration will always be decreased by a downstream movement of the outfall, but the concentration at the location of the outfall and downstream of this location may actually be increased. Consequently, if the water quality of the beaches at the mouth of the estuary are of prime consideration, an upstream location of the outfall might be preferable, though an offshore location in the coastal water would generally be even better.

The receiving capacity of a given part of the aquatic environment is related to the concentration of the pollutant that is a function of the rate of dilution and the residence time within the estuary. Because of the augmented transport in estuaries, the residence time tends to be short in comparison to that in rivers and in most lakes. In estuaries, the residence time is found by dividing the volume of freshwater within any given segment of the estuary by the rate of river flow. The freshwater fraction is calculated as illustrated in Table 6. This, multiplied by the total volume within the segment, gives the volume of freshwater. In lakes and rivers, the entire volume of water in a given part or segment is fresh, so that the total volume is divided by the rate of river flow to determine the residence time for these freshwater aquatic environments. Using the total volume in an estuary gives an erroneously long residence time, but it is sometimes done and produces a meaningless number. Another error that occurs in the evaluation of estuaries for the disposal of pollutants is the assumption that the entire volume of the tidal prism is available for dilution on each tidal cycle. This error has led to gross overestimates of the receiving capacity of estuaries and accounts for some highly polluted conditions. If this paper does nothing but prevent the perpetuation of these two errors in evaluating the receiving capacity of estuaries, it will have achieved a useful purpose.

Nitrogen in the Hudson Estuary

The excellent studies by Malone (1976, 1977) of the phytoplankton productivity in the Hudson Estuary and the New York Bight[*] provide information to illustrate the application of these basic principles of estuarine circulation. A large amount of sewage (8×10^6 m^3/day^{-1} [10×10^6 yd^3/day^{-1}]) is added to the lower reaches of the Hudson Estuary, but the phytoplankton productivity within the estuary is low in spite of the rich nutrient content. Because of high turbidity, the euphotic zone is limited to the upper 3 to 5 m (10 to 16 ft) of the water column. Garside et al. (1976) estimated that only about 10 percent of the sewage-derived inorganic nitrogen is assimilated within the estuary; the remaining 90 percent of the nitrogen is discharged through the mouth of the estuary to the New York Bight. Essentially, therefore, the dissolved inorganic nitrogen within the estuary can be considered as a conservative pollutant, and its distribution should be determined primarily by circulation and mixing.

The average river flow of the Hudson has been estimated by Ketchum et al. (1951b) to be nearly 100×10^6 m^3/day^{-1} (130×10^6 yd^3/day^{-1}) (Table 7).[†] Thus, there are 12.4 volumes of river water to dilute each volume of sewage added. The resulting concentration of dissolved inorganic nitrogen would equal 121 mg at m^{-3} as a result of dilution by river water alone. This is more than three times the inorganic nitrogen concentration of "average" seawater (Table 1) and is greatly in excess of the usual concentration in surface estuarine or coastal water.

Seawater further dilutes the sewage nutrients added to the Hudson Estuary. The expected concentration of dissolved inorganic nitrogen at any location within the estuary can be estimated from the available information on sewage input, river flow, and the observed salinity in the water. This has been done for one location within the lower Hudson Estuary and for another location offshore using an average of 12 observations throughout the year as presented by Malone (1976). The station chosen within the estuary for this analysis is south of Manhattan Island in the main channel of the Hudson Estuary. All of the sewage pollution is added upstream of this location. The annual mean fraction of freshwater there was 36 percent, and the calculated nitrogen content was 47.56 mg at m^{-3}. This calculated value is 94.2 percent of the observed annual average nitrogen content of this water. A similar computation was made for the surface water at the station just outside of the entrance to the harbor. The annual mean freshwater content there was 10 percent. Here, the calculated nitrogen content was 17.78 mg at m^{-3}, 96.8 percent of the average annual mean nitrogen content of the same water.[‡] The nitrogen content of the water is thus reduced to about half of that found in "average" seawater given in Table .1 at the harbor entrance.

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[*] The sewage has an N:P ratio of 10.14 by weight (22.46 by atoms) so that within the estuary, phosphorus is the probable limiting nutrient. In the offshore waters of the New York Bight, the N:P ratio was generally 10 or less (by atoms) so that nitrogen is the probable limiting nutrient offshore.

[†] Malone (1977) gives 12 daily estimates of river flow at the times of his observations. The average of these daily flows is 66 percent of the mean annual given by Ketchum et al. (1951), which was derived from more complete records.

[‡] The calculated nitrogen content ignores the nitrogen in the source Hudson River since Malone (1976) reported no observations in the freshwater river above the estuary. An estimate can be made assuming that the difference between the observed and calculated content is all derived from the Hudson River. Dividing the nitrogen difference by the fraction of freshwater gives for station A-3, 8.11; for station P-1, 5.80 mg at m^{-3}. These values do not seem unreasonable for the Hudson River, which receives some pollution above the estuary.

Table 7. Sewage Derived Inorganic Nitrogen ($NO_3^- + NO_2^- + NH_3$) in the Hudson River Estuary

Mean daily sewage input (10^6 m^3/day^{-1})*	8	
Mean daily nitrogen input (10^6 g N/day^{-1})*	167	
Nitrogen content of sewage (g m^{-3} = ppm)	21	
Mean river flow (10^6 m^3/day^{-1})†	99.3	
Sewage dilution	12.4:1	
Expected N in river water (g m^{-3} = ppm)	1.69	
Expected N in river water (mg at m^{-3})	121	
Nitrogen in seawater source (mg at m^{-3})§	6.31 ± 2.8	
Observations	In upper bay‡	In surface coastal water**
Annual mean salinity	20.60 ± 5.85	28.78 ± 1.59
Annual mean fraction fresh	0.36	0.10
Calculated N content (mg at m^{-3})††	47.59	17.78
Observed mean N content (mg at m^{-3})	50.51 ± 10.81	18.36 ± 16.22
Calculated as % of observed	94.2%	96.8%

*Malone, 1976, 1977.
†Ketchum, et al., 1951.
§From Malone, 1976. Observed N in the near-bottom water at Station P_1 off the entrance to New York Harbor; location 40°28.6'N, 73°54.0'W.
‡From Malone, 1976, Station A3: 40°40.3'N to 40°38.5'N; 74°0.2'18"W.
**From Malone, 1976, Station P_1 off the entrance to New York Harbor.
††Expected N in river water (120.9 mg at m^{-3}) x fraction fresh (f) plus observed N in source seawater (6.31 mg at m^{-3}) x fraction salt water (1-f).

Nutrients in the New York Bight

The pollution load of the Hudson River is only one of several sources of nutrients to the offshore waters of the New York Bight. The Raritan River pollution is added to that of the Hudson, and sewer outfalls empty directly into the Bight along the coasts of New Jersey and Long Island. There are also nutrients in the runoff both from rivers and from urban areas. Dumping at sea includes both sewage sludge and dredged spoils that are barged to the Bight for sea disposal. Mueller et al. (1976) discuss the various sources of organic carbon, nitrogen, and phosphorus to the waters of the New York Bight. Their results are summarized in Figure 2. The N:P ratio of the total inputs to the New York Bight is 9.12 (by atoms). This suggests that the phytoplankton productivity is nitrogen-limited as Malone (1977) concluded from direct observations in the Bight apex area.

In the coastal waters of the apex of the New York Bight, the sewage-introduced nutrients are rapidly assimilated by the growing phytoplankton. Malone (1976) estimated that the annual production in an area of about 600 km^2 (232 mi^2) was about 370 g C/m^{-2}, which is approximately equal to the rate of productivity in upwelling systems, the richest marine areas in the world (Ryther, 1969).* From his 12 observations throughout the year, Malone estimated that it would take between 0.4 and 10.9 days (mean = 3.22 days) for the phytoplankton to assimilate the available dissolved inorganic nitrogen in the water. He also estimated that it would take between 2.6 and 13.8 days (mean = 4.94 days) to double the standing stock of detrital carbon.

The sewage sludge disposal from barges is of particular interest in terms of this discussion since this is the material removed from the sewage in the treatment plants. Mueller et al. (1976) list the sewage sludge characteristics of 28 different plants, 12 of

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*This rate of production would require 61.4 g $N/m^{-2}/yr^{-1}$ at the normal C:N weight ratio of 6.03. For an area of 600 km^2 this would utilize 60.4 percent of the nitrogen additions to the Hudson Estuary (Table 7) or 19.2 percent of the total anthropogenic nitrogen supply to the Bight (Figure 2). The remaining nitrogen is presumably assimilated over a wider area.

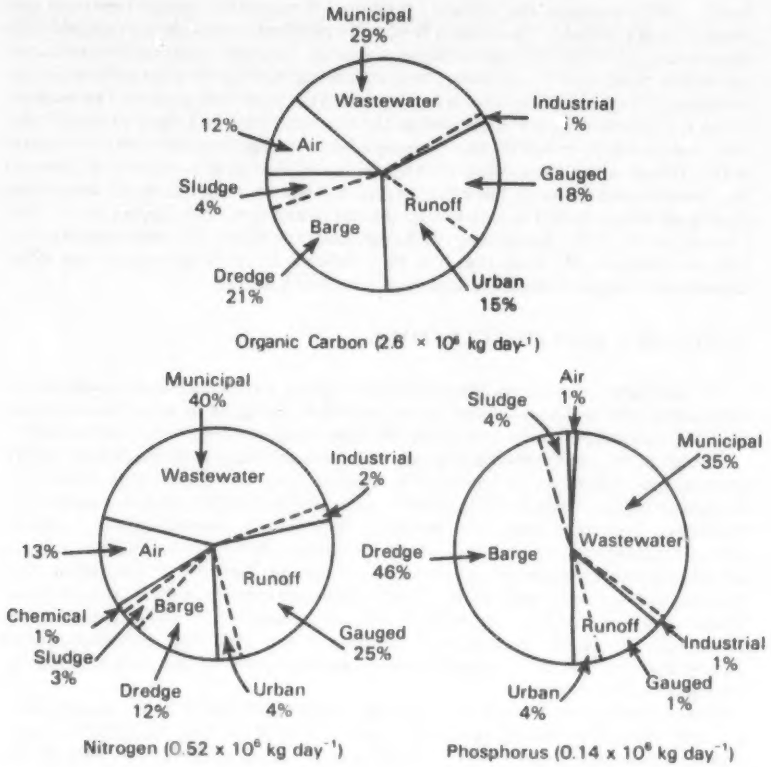

Figure 2. Proportions from different sources of carbon, nitrogen and phosphorus added to the New York Bight (after Mueller et al., 1976).

which were in New Jersey, the remainder in New York. The total mass load of nitrogen from this source, including the sources from New Jersey, is only about 10 percent of the nitrogen contributed in sewage effluents to the Hudson River. The weighted mean average concentration gives an N:P ratio of 3.75 by weight (8.30 by atoms). The sludge disposed in the Bight is, thus, more deficient in nitrogen than is the sewage effluent discharged into the Hudson Estuary. It is clear that separating the sludge from the sewage contributes little to the improvement of the water quality in the Hudson Estuary according to the data for nitrogen and phosphorus. In contrast, considerable total solids content and BOD can be removed from the sewage by separating the sludge and barging it directly to sea. This removal of organic material is clearly beneficial to the Hudson Estuary.

There is considerable controversy about the dumping of sewage sludge at sea,* and the present policy of the Environmental Protection Agency has been to phase out this type of disposal. The Environmental Protection Agency also requires that sewage receive secondary treatment prior to disposal through ocean outfalls. This necessarily creates the sludge and mandates other types of disposal. Officer and

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*The ocean dumping of dredged materials and industrial wastes is also subject to controversy, but these problems are not considered here.

Ryther (1977) compare the pollution problems of secondary sewage treatment and those of ocean outfalls. They conclude that the problems should be reexamined with appropriate scientific and engineering evaluations. Since the essential plant nutrients are largely unaffected by secondary sewage treatment, the biological cycle in the sea recreates the organic matter that was removed in the treatment process. The location of the maximum impact will depend on the characteristics and vigor of the circulation, and an offshore outfall may be preferable to sewage treatment with the release of the effluent within the estuary. Clearly, untreated sewage should not be discharged into harbors and estuaries, but offshore disposal has not been clearly shown to have significant effects except in very localized areas (Gameson, 1975; Eppley et al., 1971; Thomas et al., 1974). Sinderman (1976) discusses the effects of coastal pollution on fish and fisheries. He finds that it is very difficult to establish a cause and effect relationship except in confined and highly polluted waters.

SUMMARY AND CONCLUSIONS

The discharge of domestic pollution into coastal waters can cause beneficial or detrimental effects. Disposal operations should be designed in ways that will produce the maximum benefit and cause the least deterioration of the environment.

Sewage is frequently discharged into the confined waters of the estuary where secondary treatment for the removal of organic material is essential to reduce the biological oxygen demand of the effluent and to avoid local putrefaction and anoxia. Secondary treatment does not, however, remove the essential plant nutrients, primarily nitrogen and phosphorus, from the effluent. In the natural biological cycle, the phytoplankton assimilate these nutrients and produce an amount of organic carbon approximately equal to the amount that was removed in the treatment plant. When conditions within the estuary are not favorable for photosynthesis, the production of organic material will be delayed and maximum accumulation of organic material will be displaced downstream by a distance that is determined by the vigor of the circulation.

Within the Hudson Estuary, for example, excessive turbidity limits photosynthesis. After the polluted water reaches the coastal area of the New York Bight, phytoplankton photosynthesis is high over a wide area nourished, at least in part, by the nutrients added in pollution.

While this high production in the coastal water does not generally have a detrimental effect, wide areas of anoxia developed in the New York Bight in 1976 with associated extensive fish kills. Studies were undertaken to evaluate the cause of this event. The results have been described in a volume edited by Swanson and Sinderman (1979). Several contributing phenomena are described, including unusual meteorological conditions, an extensive bloom of the dinoflagellate *Ceratium tripos*, and high nutrients resulting in part from pollution. All may have contributed to the development of the anoxic conditions, but no single cause could be identified. The anoxic event demonstrates, however, how delicate the balance is between the high production in this area and the potential depletion of the oxygen content in the waters.

When sewage is discharged directly into coastal waters with an active circulation, it is questionable whether or not secondary treatment is desirable. The dangers of localized anoxia are minimized in coastal waters since the effluent is rapidly diluted and dispersed by the active circulation. The natural processes of the ocean serve as a sort of treatment resulting in fertilization of the marine environment and increased productivity.

The basic principles that should provide guidance in designing the discharge of sewage to the marine environment are well established. After appropriate dispersion and dilution, the nutrient concentrations in the environment should not permit the development of more organic material than can be decomposed by the available oxygen in the system. In a simple quantitative sense, these relationships are well known.

The quality of the phytoplankton population developed under different environmental conditions is, however, not yet well understood. Even under natural conditions, the species composition changes seasonally. Temperature is an important determinant of these changes. The addition of pollution can also change the normal species distribution of the phytoplankton population. This is commonly observed in heavily polluted inshore waters. Frequently, the species that grow best under polluted conditions, and consequently dominate the population, are not desirable as food for higher trophic levels. Pollution is not a "balanced" fertilizer for phytoplankton requirements since it is generally deficient in nitrogen relative to phosphorus. The effect of variable ratios of essential elements on the species composition of natural phytoplankton populations is just beginning to be understood, and much more study will be required before definite conclusions concerning the effect can be reached.

The distribution of a conservative element within an estuary can be evaluated or predicted readily for steady-state conditions as illustrated for the Hudson Estuary. The details of the distribution, and particularly the mechanisms that control and produce the observed distributions, are more difficult to understand and predict. More sophisticated models are necessary for this purpose. They are particularly valuable to evaluate transient events and to assess the effects of biological or other time-variable processes in producing the observed distributions. It is gratifying that progress is being made to achieve an understanding of these problems, but much remains to be done.

In conclusion, it seems appropriate to make use of the assimilative capacity of marine coastal waters for the disposal of the wastes of human metabolism. This can be done properly only by making the best use of scientific understanding of the entire system, including the biological, physical, and chemical characteristics. Mistakes have been made in the past, partly because the system as a whole was not adequately understood. Today, there is no excuse to repeat the mistakes of the past, and it should be possible to design disposal operations to achieve the maximum benefit without detrimental effects.

REFERENCES

Doig, M.T., and D.F. Martin. 1974. The response of *Gymnodinium breve* to municipal waste materials. Mar. Biol. 24:223–228.

Eppley, R.W., A.F. Carlucci, O. Holm-Hansen, D. Kiefer, J.S. McCarthy, E. Venrick, and P.M. Williams. 1971. Phytoplankton growth and composition in shipboard cultures supplied with nitrate, ammonium or urea as the nitrogen source. Am. Soc. Limnol. Oceanogr. 16:741–751.

Fuhs, G.W., S.D. Demmerle, E. Canelli, and M. Chen. 1972. Characterization of phosphorus-limited plankton algae (with reflections on the limiting-nutrient concept). Am. Soc. Limnol. Oceanogr. Spec. Symp. 1. pp. 113–133.

Gameson, A.L.H., ed. 1975. Discharge of sewage from sea outfalls. Pergamon Press, Elmsford, NY.

Garside, C., T.C. Malone, O.A. Roels, and B.C. Sharfstein. 1976. An evaluation of sewage-derived nutrients and their influences on the Hudson Estuary and the New York Bight. Estuarine and Coastal Marine Science 4:281–289.

Goldman, J.C. 1976. Identification of nitrogen as a growth limiting nutrient in waste waters and coastal marine waters through continuous culture algal assays. Water Research 10:97–104.

Goldman, J.C. 1979. Temperature effects on steady-state growth, phosphorus uptake and the chemical composition of a marine phytoplankter. Microbial Ecology 5:153–166.

Goldman, J.C., J.J. McCarthy, and D.G. Peavey. 1979. Growth rate influence on the chemical composition of phytoplankton in oceanic waters. Nature 279(5710):210–215.

Ketchum, B.H. 1939. The development and restoration of deficiencies in the phosphorus and nitrogen composition of unicellular plants. J. Cell. Comp. Physiol. 13(3):373–381.

Ketchum, B.H. 1951a. The exchanges of fresh and salt waters in tidal estuaries. Jour. Mar. Res. 10:18–38.

Ketchum, B.H. 1951b. The flushing of tidal estuaries. Sewage and Industr. Wastes 23:198–209.

Ketchum, B.H. 1955. Distribution of coliform bacteria and other pollutants in tidal estuaries. Sewage and Industr. Wastes 27:1288–1296.

Ketchum, B.H. 1969. Eutrophication of estuaries. In: Eutrophication: causes, consequences, correctives. Proceedings of a Symposium, National Academy of Sciences, Washington, D.C. pp. 197–209.

Ketchum, B.H. 1973. Ecological effects of marine pollution. Special Publication, Mar. Biol. Assn. India. pp. 7–28.

Ketchum, B.H., J.C. Ayers, and R.F. Vaccaro. 1952. Processes contributing to the decrease of coliform bacteria in a tidal estuary. Ecology 33:247–258.

Ketchum, B.H., A.C. Redfield, and J.C. Ayers. 1951. The oceanography of the New York Bight. Papers in Physical Oceanography and Meteorology 12(1).

Ketchum, B.H., R.F. Vaccaro, and N. Corwin. 1958. The annual cycle of phosphorus and nitrogen in New England coastal waters. Jour. Mar. Res. 17:282–301.

Malone, T.C. 1976. Phytoplankton productivity in the apex of the New York Bight. U.S. Department of Commerce, Marine Ecosystems Analysis Program. Boulder, CO. NOAA Technical Memorandum ERL, MESA-5.

Malone, T.C. 1977. Environmental regulation of phytoplankton productivity in the lower Hudson Estuary. Estuarine and Coastal Marine Science 5:157–171.

Mancini, J., J.A. Mueller, J.J. Roswell, J. Goldman, A.I. Mytelka, T. Bober, E. Myers, and E. Tobier. In press. Urban discharges. U.S. Department of Commerce, NOAA Special Report Series on Perspectives of the New York Bight.

Miller, W.E., J.C. Greene, and T. Shiroyama. 1975. Application of algal assays to define the effects of wastewater effluents upon algal growth in multiple use river systems. In: Biostimulation and nutrient assessment symposium proceedings, E.J. Middlebrooks, D.H. Falkenborg and T.E. Maloney, eds. PRWG 168-1, Logan, UT.

Mueller, J.A., J.S. Jerris, A.R. Anderson, and C.F. Hughes. 1976. Contaminant inputs to the New York Bight. U.S. Department of Commerce, NOAA Technical Memorandum ERL, MESA-6.

O'Connor, D.J. In press. Modeling of eutrophication in estuaries. In: Enrichment of estuaries, B.J. Neilson,and L.E. Cronin, eds: Proceedings of the International Symposium, May 29–31, 1979, Williamsburg, VA. Chesapeake Research Consortium, Humana Press, Inc., Clifton, NJ.

O'Connor, D.J., and D.J. Thomann. 1971. Water quality models: chemical, physical and biological constituents. In: Estuarine modelling: an assessment. G.H. Ward, Jr., and W.H. Espey, Jr., eds. Water Pollution Control Research Series, Water Quality Office, U.S. Environmental Protection Agency, Washington, DC. pp. 102-168.

Officer, C.B., and J.H. Ryther. 1977. Secondary sewage treatment versus ocean outfalls: an assessment. Science 197:1056–1060.

Perry, M.J. 1976. Phosphate utilization by an ocean diatom in phosphorus-limited chemostat culture and in the oligotrophic waters of the central North Pacific. Am. Soc. Limnol. Oceanogr. 21:88–107.

Pritchard, D.W. 1969. Dispersion and flushing of pollutants in estuaries. Jour. Hydraulics Div., Proc. Amer. Soc. Civil Eng. 95:115–124.

Redfield, A.C., B.H. Ketchum, and F.A. Richards. 1963. The influence of organisms on the composition of seawater. In: The sea, vol. 2, M.N. Hill, ed. Interscience, NY. pp. 26–77.

Ryther, J.H. 1969. Photosynthesis and fish production in the sea. Science 166:72–76.

Ryther, J.H., and W.M. Dunstan. 1971. Nitrogen, phosphorus and eutrophication in the coastal marine environment. Science 171:1008–1013.

Segar, D. In press. Contamination of populated estuaries and adjacent coastal ocean—a global review. U.S. Department of Commerce, NOAA Special Report Series on Perspectives of the New York Bight.

Sinderman, C.J. 1976. Effects of coastal pollution on fish and fisheries—with particular reference to the Middle Atlantic Bight. In: Middle Atlantic Continental Shelf and the New York Bight, M.G. Gross, ed. Am. Soc. Limnol. Oceanogr. Spec. Symp. 2. Allen Press, Lawrence, KS. pp. 281–301.

Swanson, R.L., and C.J. Sinderman, eds. 1979. Oxygen depletion and associated benthic mortalities in New York Bight. NOAA Professional Paper 11. U.S. Government Printing Office.

Thomas, W.H., D.L.R. Seibert, and A.N. Dodson. 1974. Phytoplankton enrichment experiments and bioassays in natural coastal seawater and in sewage outfall receiving waters off southern California. Estuarine and Coastal Marine Science 2:191–206.

Tully, J.P. 1949. Oceanography and prediction of pulp mill pollution in Alberni Inlet. Bull. Fish. Res. Bd. Canada 83.

U.S. Environmental Protection Agency. 1974. Marine algal assay procedure bottle test. National Environmental Research Center, Corvallis, OR. U.S. Government Printing Office: 1975-697-829.

IMPACT OF TOXIC ORGANICS ON THE COASTAL ENVIRONMENT

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The variety of toxic organic compounds being introduced into estuarine and coastal waters of the United States and the extent to which they may affect the environment present awesome considerations for the eighties. Most of the compounds described as toxic to estuarine organisms fall within five general categories: herbicides, fungicides, pesticides, petrochemicals, and industrial compounds or their byproducts. Although some of the effects of both herbicides and fungicides on marine organisms have been investigated, there is little indication to date that these compounds represent a serious threat to the estuarine environment. Many toxic organic compounds are associated with petroleum and its refined products, but, as with the industrial compounds, these organics do not constitute a continuous and long-term threat because they are not routinely used in the coastal environment. Many toxic organic compounds are also associated with industrial processes and, in some cases, the polychlorinated biphenyls (PCBs), for example, are introduced into the estuarine environment through carelessness or by accident rather than through routine use in areas adjacent to tidal waters, as with pesticides.

This review will focus largely on the impact of pesticides and on certain industrial products. It will consider the extent to which some of these compounds have a deleterious effect on estuarine and marine organisms. It will also identify those aspects of the basic problem of impact that have not been adequately addressed in light of the emerging problems, those associated with further development and future use of increasingly sophisticated compounds.

At the beginning of the seventies, individual scientists as well as state and federal agencies began to be concerned with how the continuous addition of pesticides to estuarine and marine ecosystems might affect individual species, the marine ecosystem as a whole, and humankind as well. The banning of the use of DDT in the United States in 1972 led to the development and use of many new compounds. Primarily, these were more specific in their action and thought to be less persistent in either the marine environment or in biological systems. The majority of these compounds were organochlorine or organophosphate compounds. Publications throughout the decade have reported the effects on marine organisms of several of the organochlorine compounds including aldrin, dieldrin, mirex, methoxychlor, and Kepone. Virtually all of the organochlorine insecticides act on the nervous system. Less scientific interest has been demonstrated in the effects of the organophosphates, compounds that apparently inhibit the action of enzymes. This group includes parathion, malathion, and diazinon. The effectiveness of several of the newer organophosphate pesticides stimulated the development of still other compounds designed to inhibit enzyme activity, including the carbamates and such compounds as diflubenzuron (Dimilin®), which inhibits the normal secretion of cuticle in arthropods. Concurrently, a group of insecticides identified as third generation pesticides were developed by the chemical industry. These either are insect hormones or mimic the action of insect hormones in such a way as to prevent growth or metamorphosis.

There appears to be a reduction in the number of compounds that are broadly toxic to biological systems and an increase in the number of compounds developed specifically to inhibit enzymatic or hormonal reactions in insects. This development frequently proceeded without consideration of the extent to which these same enzyme and hormonal systems are found in arthropods in general, including virtually all of the marine Crustacea, and what effect these compounds might therefore have on those nontarget arthropods closely related to the insects. In addition to those marine Crustacea that are commercially important—the penaeid shrimps and a number of species of crabs, for example—the commercial finfisheries of the United States depend on a variety and abundance of marine Crustacea found within the planktonic food web. Nothing is known about the effects of the more specific compounds available at the beginning of the eighties on food web Crustacea. The extent to which many of these compounds may be accumulated through the food chain has been identified for a relatively small number of compounds.

GENERAL EFFECTS OF TOXIC ORGANICS ON ESTUARINE BIOLOGICAL SYSTEMS

A considerable number of scientific publications, agency reports, and periodic reviews on effects of toxic organics have appeared during the seventies (Walsh, 1972). In fact, the emergence of literature dealing with environmental pollution from heavy metals, organics, and petrochemical compounds has led to the development of specific scientific journals devoted to these topics. Papers have dealt with sublethal and lethal effects of specific compounds on individual species; behavioral and physiological responses to specific compounds; bioaccumulation and residual amounts of organic compounds in estuarine animals, plants, and sediments; and a consideration of adequate methods to analyze and evaluate the impact of toxic materials in estuarine and marine waters.

Pesticides

Although the lethal effects on marine fish, crabs, and molluscs of many organic compounds used as pesticides are generally well documented, the number of compounds that have been examined in detail for their toxic effects on each of these groups is extremely small. In studies involving post-larvae of differing species, there appears to be considerable variation in the levels of toxicity of many of the compounds. This is in part due to the physiological state or stage of development of the organisms under study. For example, juvenile *Cancer magister* exposed to 4.0 mg/l of methoxychlor died within 72 hours after application, whereas adult crabs of the same species died only when exposed to 40.0 mg/l (Armstrong et al., 1976). This level of toxicity in Dungeness crabs is similar to that described for DDT (Poole and Willis, 1970), and the decline in landings of this species in San Francisco has been attributed to the increased use of pesticides in that area. Williams and Duke (1979) summarized much of the available literature on the effects of many chlorinated hydrocarbons, organophosphates, carbamates, juvenile hormone mimics, and insect growth regulators on estuarine Crustacea. They concluded that such compounds as dieldrin, DDT and its metabolites, Kepone, mirex, malathion, and carbofuran almost always have toxic effects on adult blue crabs, fiddler crabs, and estuarine shrimps.

Lethal levels for many planktonic organisms, including the larvae of marine Crustacea, have also been documented. Epifanio (1979) considered the work of several recently published authors in arriving at an understanding of how chlorinated hydrocarbons, organophosphates, carbamates, and juvenile hormone mimics affect the survival of larvae of many estuarine crabs. The relatively limited number of observations, most of which deal with the effects of chlorinated hydrocarbons, would suggest that all have toxic effects on one or more of the larval stages. In certain experiments, such as those describing the effect of juvenile hormone mimics on larvae of *Rhithropanopeus harrisii*, specific stages during larval development seem more sensitive than others (Costlow, 1977). To test the degree to which individual larval stages differ in levels of sensitivity to methoprene, larvae were treated in two ways. In one series, larvae were maintained in three salinities combined with 0.1 ppm methoprene for specific periods of time and then removed to seawater without methoprene for the remainder of their development. In another, they were initially maintained in water without the methoprene and then moved to water containing 0.1 ppm methoprene for the remainder of their larval and early juvenile lives. In both instances, at the optimum salinity of 20‰, relatively little mortality was observed until day 13, which corresponded with the completion of the fourth zoeal molt to the megalopal stage. None of the megalopa molting from zoea maintained under these two conditions survived the final metamorphic molt to the first juvenile stage (Costlow, 1977). As with studies on adult animals, however, levels of toxicity appear to vary considerably, depending largely upon the species and the natural conditions of salinity and temperature.

Many sublethal effects have also been identified for adult and larval estuarine organisms, but the full implications of these studies are not totally understood. For example, one juvenile hormone mimic, initially developed to inhibit the metamorphosis of larval insects, has been shown to affect the reproductive cycle of one species of intertidal crab (Payen and Costlow, 1977).

Studies on the effects of methoxychlor on several species of crustacean larvae demonstrate several sublethal effects. Although exposures of up to 10 μg/l of methoxychlor did not reduce the percentage of *Cancer magister* eggs hatching within 24 hours, successful development to the extremely short prezoea stage decreased over that observed in the controls (Armstrong et al., 1976). Only 70 percent of hatched organisms successfully molted to the first true zoea. A reduction in motility was observed in 50 to 90 percent of those that did develop when they were reared in concentrations of methoxychlor ranging from 0.18 to 1.0 μg/l.

Concentrations of Dimilin® ranging from 1 to 10 ppb did not affect the survival of adult copepods *(Acartia tonsa)* but did alter the viability of eggs produced by females maintained in this insect growth regulator (Tester and Costlow, 1979). When females were maintained in 10 ppb Dimilin® for periods ranging from 12 to 36 hours, the percentage of eggs hatching decreased from 93.4 to 1.2. Concentrations of 1 ppb Dimilin® also reduced egg viability, an effect that was most pronounced following 36 to 60 hours of treatment. Those eggs that did not hatch were fully developed and appeared to be viable, with nauplii observed moving within the egg membranes. The nauplii that did hatch later in the treatment period were of abnormal shape and failed to molt to the second naupliar stage. Most frequently, body and appendage shape and setae were found to be abnormal (Tester and Costlow, 1979).

Juveniles of *Cancer magister* exposed to methoxychlor were found to be smaller than crabs within the control series. Sensitivity of juvenile crabs to this compound appear to be greatest during the period of ecdysis or shortly thereafter. Adult crabs exposed to methoxychlor were hyperactive, exhibiting much more frequent movement of mouthparts and chelipeds than observed in the control animals. The more severely affected crabs were incapable of maintaining an upright posture, and some remained supine for several weeks before death. Crabs exposed to 40 μg/l of methoxychlor ceased to eat, whereas crabs exposed to considerably lower levels appeared to have difficulty in locating food. They were also observed to tear at the food, leaving it scattered within the container rather than ingesting it.

Similar behavioral changes associated with the ingestion of small amounts of pesticide have been recorded. After exposure to DDT, fiddler crabs exhibited a variety of behavioral irregularities (Odum et al., 1969), and Klein and Lincer (1974) observed similar behavioral changes after *Uca pugilator* had ingested small amounts of dieldrin. After exposure to 10 ppm dieldrin, the adult crabs were unable to right themselves; when the concentration was reduced to 1 ppm, the righting response was measurably delayed. Modifications in the speed and pattern of attempts to escape were also observed. Crabs exposed to 1 to 50 ppm of dieldrin developed sluggish movements. At the highest concentrations, the normal response, that of raising the major cheliped, frequently caused the crabs to fall over backwards, after which they were unable to right themselves. The ability of adult crabs to coordinate properly in both the righting response and in coordinated escape behavior is essential for survival, because both responses enable the crab to avoid predation and dehydration.

The response of planktonic organisms to light is affected by trace amounts of several pesticides (Forward and Costlow, 1976). Sublethal concentrations of methoprene, a synthetic juvenile hormone mimic, had no effect upon either swimming speed or phototaxis of larvae of the mudcrab, *Rhithropanopeus harrisii*. Hydroprene, another juvenile hormone mimic, had no significant effect upon swimming speed during the first three zoeal stages but produced a marked increase in swimming speed in the stage-four zoea. Phototaxis was almost entirely unaltered by exposure to hydroprene; however, stage-three zoea maintained at the highest concentration of the compound (0.1 ppm) demonstrated a significantly increased level of positive phototaxis. Whereas this level was higher than that observed for the control larvae, it was significantly lower than that observed for the larvae tested in the acetone control, the carrier employed to maintain hydroprene in solution (Forward and Costlow, 1978). A third compound, Dimilin®, produced a pronounced effect on the larvae of *Rhithropanopeus harrisii*, even when sublethal levels of the compound were used. Although swimming speeds were generally accelerated, this response varied considerably from one larval stage to the other. Phototaxis was unaffected within the first three zoeal stages; in stage four, however, although swimming patterns were normal, phototactic response was drastically altered. Swimming and phototactic responses enable the organism to avoid predators and adverse salinity and temperature conditions. Phototaxis probably aids the organism in maintaining the proper vertical position for horizontal migrations (Bousfield, 1955). Any variation in these basic behavioral patterns could therefore impair survival even though mortality may not be attributed directly to the compounds in question. The results of these studies suggest that this approach may be a most useful bioassay to determine the presence of extremely small amounts of pollutants in estuarine waters.

Sublethal effects of methoxychlor have also been observed at the physiological level. Caldwell (1974) describes the way in which methoxychlor affects osmotic and ionic regulation in two species of adult crabs, *Cancer magister* and *Hemigrapsus nudus*. Sublethal levels resulted in a decreased resistance in tolerance of the adult crab to reduced salinity and also resulted in partial inhibition of the gill NaK mg ATPases in *Cancer magister*. The author, however, could not demonstrate that osmotic regulation in *Hemigrapsus nudus* or osmotic and ionic regulation in *Cancer magister* were significantly impaired by these treatments.

The duration of individual larval stages, long known to be affected by temperature and salinity, has been further shown to be affected by the presence of certain compounds in the water column. Increased concentrations of Kepone, methoxychlor, and malathion resulted in reduced molting rates in the mud crab *Rhithropanopeus harrisii* and the blue crab *Callinectes sapidus* (Bookhout et al., 1976, 1980; Bookhout and Monroe, 1977). Whereas Dimilin® was toxic during larval stages of several species of estuarine crab *(Rhithropanopeus harrisii* and *Sesarma reticulatum)*, it did not affect the duration of the larval development (Christiansen et al., 1978). At concentrations of 1.0 to 5.0 ppb, the duration of zoeal development of the mud crab *Rhithropanopeus harrisii* and the blue crab *Callinectes sapidus* was increased as the concentrations of methoxychlor were increased (Bookhout et al., 1976).

Morphological abnormalities or the appearance of extra or supernumerary larval stages have also been observed. In the stone crab, *Menippe mercenaria*, exposure to mirex caused no significant increase in duration of developmental stages, but the percentage of extra sixth zoeal stages increased as the concentration of mirex increased from 0.01 to 1.0 ppb. The majority of the sixth zoeal stages died before reaching the megalopa stage (Bookhout et al., 1972). In similar studies on larvae of the blue crab, *Callinectes sapidus*, however, there was no indication that large numbers of the eighth zoeal stage occurred when the larvae were exposed to higher concentrations of mirex (Bookhout and Costlow, 1975).

Studies on the effect of methoprene on the larvae of the mud crab, *Rhithropanopeus harrisii*, indicated that while a variety of concentrations of this juvenile hormone mimic did not increase mortality, the combination of high salinity (35 ppt) and 0.1 ppm methoprene resulted in increases in morphological abnormalities of the megalopa (Costlow, 1977). In lower salinities (5 ppt and 20 ppt), abnormal megalopa rarely exceeded 4 percent, regardless of the length of an exposure to the compound. Although some abnormal megalopa successfully metamorphosed to the first juvenile crab, many died either as megalopa or during metamorphosis to the first crab.

Toxic Industrial Compounds

Many other industrial compounds developed since the late 1920s have been identified in estuarine sediments and waters over much of the United States. Some, such as the phenolics, are derived from coke plants, oil refineries, chemical and pesticide manufacturers, and other industrial complexes. Many of the phenolics occur naturally in aquatic and terrestrial vegetation and may be released as a result of processes employed by the pulp and paper industry. Buikema, McGinniss, and Cairns (1979) identify various toxic effects for some of the phenolics. The few studies conducted on the biological effects of many phenolics indicate that exposure to phenol and pentachlorophenol concentrations as low as 4 mg/l results in hemorrhaging at the base of the fins of fish, and higher concentrations cause disruption of blood vessel walls and gill epithelium. Other effects include the reduction of levels of the hormone in fish, changes in blood glucose and blood lactate levels, immunoglobin levels, blood protein levels, and tissue microelement levels. Little is known about the cycling of phenol and phenolics (other than pesticides) in marine ecosystems or the extent to which they persist in substrates and biological systems.

The polychlorinated biphenyls (PCBs), one of the groups of chlorinated hydrocarbons, are widely used in condensor dielectrics, heat transfer fluids, and hydraulic fluids. They are widely distributed in marine and estuarine environments, and the work of Peakall (1975) established that they are toxic to many organisms. The polychlorinated naphthalenes have also been identified in the marine environment, but relatively little is known about the effect of these compounds on estuarine and marine systems. Many PCBs have been produced by Monsanto Company under the trade name Aroclor®. Under the trade name Halowax®, the Koppers Company has produced a series of polychlorinated naphthalenes. A number of studies have been conducted to compare the effects of the PCBs and the PCNs within these series.

Following the identification of Aroclor® 1254 in the water, sediment, and fauna of Escambia Bay, Florida (Duke et al., 1970), Nimmo conducted several studies on the effects of this and related compounds on various estuarine and marine animals. Whole body residues of Aroclor® 1254 were found to be as high as 14 mg/kg in the pink shrimp *Penaeus duorarum* (Nimmo et al., 1971a), and subsequent studies on juveniles of the same species indicated that approximately 1.0 μg/l in seawater would kill 50 percent of the experimental animals within 15 days (Nimmo et al., 1971b).

Lethal effects of Aroclor® 1016 and 1254 have been described for larval and adult fiddler crabs using a combination of temperatures and salinities as synergistic factors (Vernberg et al., 1977). Lethal levels of Aroclor® 1254 for the larval stages of the fiddler crab were found to be approximately 10 ppb, but the same levels of Aroclor® 1254 appeared to have a more rapid effect on the larvae than that described for Aroclor® 1016. At the combinations of salinity and temperature used in the experiments, Aroclor® 1254 appeared to be more toxic than Aroclor® 1016.

Sublethal effects of PCBs have also been described for many species. In the studies on *Palaemonetes pugio*, Roesijadi et al. (1976) observed sublethal effects in concentrations of less than 100 ppb. Neff and Giam (1977) found that lower concentrations of Aroclor® and Halowax® reduced the intermolt periods of the horseshoe crab, while Roesijadi et al. (1976) demonstrated that Aroclor® 1254 significantly extended the duration of development of the shrimp *Palaemonetes pugio*. In describing the relationship between pollutants and natural diseases Couch and Courtney (1977) found that 1 to 3 µg/l Aroclor® 1254 administered for 30 days increased the spread and prevalence of *Baculovirus*. Mortality in the stress population was higher than in the control population, and the incidence of viral infection in the experimental population of shrimp was approximately 50 percent higher than in the control series without Aroclor® 1254.

Neff and Giam (1977) conducted studies to compare the effects of Aroclor® 1016 and the PCN Halowax® 1099 on juvenile horseshoe crabs, *Limulus polyphemus*. The concentrations of these compounds, while in the low ppb range, were considerably higher than one would expect to find in most estuarine environments.

The anticipated effects of "hazardous wastes" constitute a major challenge. Numerous documents attest to the increase in production of hazardous wastes within the United States through the year 2000. Several recent incidents involving contamination of drinking water, residences, and the environment as a whole have dramatically focused on the hazards of these compounds to human health. Frequently the compounds are unidentified and assembled in what might be best described as a "potpourri." Although the Environmental Protection Agency is presently conducting studies to devise means to identify these hazardous wastes, there is virtually no way to assess the impact from runoff of these compounds into adjacent estuarine waters.

THE CHALLENGES OF THE EIGHTIES

Although research during the seventies has contributed to a general understanding of the effects of some organic compounds on estuarine and marine organisms, many questions about the long-term impacts of these compounds are unanswered. Little information is available on the rates at which many of these compounds deteriorate within estuarine waters and sediments, although comparable information on rates of deterioration in freshwater and soil is available. The results of one study (Christiansen and Costlow, 1980) show that Dimilin® breaks down relatively slowly in brackish water. Under laboratory conditions, 10 ppb Dimilin®, added to seawater, degraded for 8 weeks before it had reached a level that did not affect survival of the larvae of the mud crab *Rhithropanopeus harrisii*.

Although the breakdown products of specific organic compounds have been identified, virtually nothing is known of the level at which these compounds are toxic in estuarine systems, the extent to which they persist, or the amounts that may be accumulated by marine organisms and passed on through the various trophic levels.

Research in the seventies concentrated on the effects of individual compounds on relatively small numbers of estuarine species. Although toxicity of individual pollutants to marine organisms is modified by synergistic factors including salinity and temperature, little research has been directed to a thorough understanding of these interactions. The mortality of shrimp exposed to Aroclor® 1254 increased as salinity increased over an 8-hour period. In contrast, mortality did not increase when shrimp were maintained in conditions where only the salinity was increased or where only PCB was present (Nimmo and Bahner, 1974). In studies on the effects of two juvenile hormone mimics (Altosid® and Altozar®) on larval development of mud crabs, a variety of cyclic temperatures were used to indicate that the toxic effect of these two compounds is reinforced by temperature (Christiansen et al., 1977).

Considering the extent to which estuarine and coastal waters contribute runoff containing many organic compounds, it is unfortunate that most studies to date have focused on the effects of single pollutants rather than on how combinations of pollutants alter toxic effects. Livingston et al. (1974) considered these interactions, but Bahner and Nimmo (1976) first examined how combinations of heavy metals and pesticides (cadmium-malathion; cadmium-methoxychlor; and cadmium-methoxychlor-Aroclor® 1254) might affect the pink shrimp, *Penaeus duorarum*. They concluded that the absence of cumulative toxicity indicated that toxicity is independent and that no synergistic activity occurred.

Koenig (1977) developed experiments to determine the relative toxicity of DDT and mirex, alone as well as in combination, during the early life stages of the salt marsh cyprinodont fish, *Adinia xenica*. He identified a synergistic interaction between DDT and mirex that affected larval mortality. The addition of mirex to the DDT dosages increased the toxicity by a factor of approximately 1.5. There was no apparent synergistic effect on embryos, embryo developmental rate, or hatching time. Secondary effects of DDT intoxication, as opposed to the effects of mirex alone, included lack of coordination, cessation in feeding, and increased darkening in overall appearance. The larvae soon became emaciated and died. Larvae exposed to mirex, however, fed normally and initially appeared normal. After a loss of equilibrium, which caused the larvae to drift in a disoriented manner, death occurred.

In more recent studies (Costlow, unpublished) four pesticides, Dimilin®, Altosid®, mirex, and Kepone, were combined at levels previously determined to be sublethal. Developing stages of the mud crab *Rhithropanopeus harrisii* were exposed to sublethal levels both of the individual pesticide and a composite of the four. At the sublethal levels of the individual compounds, survival was similar to that observed for the controls, whereas for the composites, toxic effects increased considerably. Although the previous studies were conducted under conditions approaching optimum salinity and temperature, further experiments are necessary to determine the effects of composites of pollutants in those suboptimal conditions of salinity and temperature known to occur in estuarine systems.

Virtually all of the research of the seventies was directed to acute effects. A few studies sought to evaluate chronic effects, primarily as they apply to assemblages of organisms. Livingston et al. (1978) described a significant decline in organochlorine residues in Apalachicola Bay, Florida. They attributed this decline to a decrease in use of pollutants, a major natural flushing of the area, and the deterioration of the compounds themselves. A major problem in this study was the difficulty in correlating trends in the various assemblages with the decline of the pesticides. The need for studies that relate long-term changes and physical-chemical changes, be they natural or artificial, is therefore emphasized. This same difficulty is apparent in other efforts to relate the decline in species abundance to the observed increase of pesticides in specific estuarine waters. For example, the recorded commercial catch of blue crabs in the James River from 1968 to 1972 averaged 899,000 kg (1,977,800 lb) and declined from 1972 to 1975 by more than 90 percent. In noting this decline Bookhout et al. (1980) indicates that Kepone, shown to be highly toxic to the larvae of *Callinectes sapidus*, may be responsible for the decline of the blue crab catch in the James River. There is, however, no way to specifically attribute this decline to Kepone poisoning alone.

Virtually nothing is known of the mutagenic effects of organic compounds over several generations of estuarine and marine organisms. Techniques are now available to permit the culture of many invertebrates, including certain harpacticoid copepods and polychaetes that have relatively short life spans (10 to 20 days). For a more complete understanding of the long-term effects of pollutants in estuarine ecosystems, studies must be designed to determine how sublethal levels of the more commonly utilized organic compounds affect successive generations of several species.

Although many studies have been conducted on the extent to which individual species accumulate organic compounds, nothing is known about the mechanism of uptake, transfer within tissues and organs, and the mechanisms that permit certain organisms to accumulate these compounds without apparent toxic effects.

Despite significant advances in instrumentation during the seventies, researchers are still seriously handicapped by their inability to detect and accurately measure small amounts of organic compounds in the estuarine environment. Virtually all experimental studies on the impact of organics on estuarine species have been conducted under laboratory conditions using either static or flow-through systems that rely on conventional dilution techniques to arrive at the lower levels of concentration. While laboratory studies establish basic physiological responses of individual species to particular compounds, laboratory conditions do little more than simulate the natural environment. Reliable information is still unavailable either on the levels of compounds in estuarine systems or on how these levels affect a variety of biological responses within the organisms.

The significance of the impact of toxic organic compounds on the estuarine and marine environments cannot be overestimated, because these areas provide protein for the expanding populations of the world. The 1976 FAO World Conference on Aquaculture concluded that a five- to ten-fold increase in production of fisheries products from aquaculture would be possible by the year 2000, given adequate financial and technical support. Although limited investment and technical support are now making such an increase unlikely, increasing pollution of freshwater ponds and coastal waters are a far more serious threat (Barney, 1980). The expanded use of pesticides is expected to increase water pollution in many of the developing countries, where the more persistent pesticides are likely to be in continued use. Pesticide use in the developing countries may well quadruple between now and the year 2000 (Barney, 1980). According to the Global 2000 Study, pollution of coastal ecosystems is likely to increase. The study indicates that 60 to 80 percent of the valuable commercial marine species are dependent upon estuaries, salt marshes, or mangrove swamps for a habitat during their life cycle. As cities expand and industries develop estuarine and coastal wetland areas, the fragile environments on which these commercially important species depend will deteriorate.

Perhaps the greatest challenge of the eighties will be to communicate to local governments in a number of countries the importance of estuarine and coastal areas and how toxic organic compounds can destroy their productivity. Currently, there appears to be little concern, despite the 1972 Coastal Zone Management Act and coastal management laws later enacted by a number of states. Without proper recognition of the impact of toxic organic compounds on the productivity of coastal areas by local and state governments, accompanied by intelligent planning that prevents further contamination of these areas, the prospect for continued productivity of these areas is extremely dim.

REFERENCES

Armstrong, D.A., D.V. Buchanan, M.H. Mallon, R.S. Caldwell, and R.E. Millemann. 1976. Toxicity of the insecticide methoxychlor to the Dungeness crab *Cancer magister*. Mar. Biol. 38:239–252.

Bahner, L.H., and D.R. Nimmo. 1976. Metals, pesticides, and PCB's: toxicities to shrimp singly and in combination. In: Estuarine processes, vol. 1., M. Wiley, ed. Academic Press, NY. pp. 523–532.

Barney, G.O. 1980. The global 2000 report to the president: entering the twenty-first century. A report prepared by the Council on Environmental Quality and the Department of State, vol. 1.

Bookhout, C.G., and J.D. Costlow, Jr. 1975. Effects of mirex on the larval development of blue crab. Water, Air, and Soil Pollution 4:113–126.

Bookhout, C.G., J.D. Costlow, Jr., and R. Monroe. 1976. Effects of methoxychlor on larval development of mud crab and blue crab. Water, Air, and Soil Pollution 5:349–365.

Bookhout, C.G., J.D. Costlow, Jr., and R. Monroe. 1980. Kepone effects on larval development of mud crab and blue crab. Water, Air, and Soil Pollution 13:57–77.

Bookhout, C.G., A.J. Wilson, Jr., T.W. Duke, and J.I. Lowe. 1972. Effects of mirex on the larval development of two crabs. Water, Air, and Soil Pollution 1:165-180.

Bousfield, E.L. 1955. Ecological control of the occurrence of barnacles in the Miramichi Estuary. Natl. Mus. Can. Bull. 137:1-69.

Buikema, A.L., Jr., M.J. McGinniss, and J. Cairns, Jr. 1979. Phenolics in aquatic ecosystems: a selected review of recent literature. Mar. Environ. Res. 2:87-181.

Caldwell, R.S. 1974. Osmotic and ionic regulation in decapod Crustacea exposed to methoxychlor. In: Pollution and physiology of marine organisms. F.J. Vernberg and W.B. Vernberg. Academic Press, NY. pp. 197-224.

Christiansen, M.E., and J.D. Costlow, Jr. In press. Persistence of the insect growth regulator Dimilin® in brackish water: a laboratory evaluation using larvae of an estuarine crab as indicator. XIVth European Marine Biology Symposium, September 1979, Helgoland.

Christiansen, M.E., J.D. Costlow, Jr., and R.J. Monroe. 1977. Effects on the juvenile hormone mimic ZR-515 (Altosid®) on larval development of the mud crab *Rhithropanopeus harrisii* in various salinities and cyclic temperatures. Mar. Biol. 39:269-279.

Christiansen, M.E., J.D. Costlow, Jr., and R.J. Monroe. 1978. Effects of the insect growth regulator Dimilin® (TH 6040) on larval development of two estuarine crabs. Mar. Biol. 50:29-36.

Costlow, J.D., Jr. 1977. The effect of juvenile hormone mimics on development of the mud crab, *Rhithropanopeus harrisii* (Gould). In: Physiological responses of marine biota to pollutants. Academic Press, NY. pp. 439-457.

Couch, J.A., and L. Courtney. 1977. Interaction of chemical pollutants and a virus in a crustacean: a novel bioassay system. Ann. N.Y. Acad. Sci. 298:497-504.

Duke, T.W., J.I. Lowe, and A.J. Wilson, Jr. 1970. A polychlorinated biphenyl (Aroclor® 1254) in the water, sediment, and biota of Escambia Bay, Florida. Bull. Environ. Contam. and Toxicol. 5:171-180.

Epifanio, C.E., 1979. Chapter VIII, larval decapods (Arthropoda, Crustacea, Decapoda). In: Pollution ecology of estuarine invertebrates. C.W. Hart, Jr., and S.L.H. Fuller. Academic Press, NY. pp. 259-292.

Forward, R.B., Jr., and J.D. Costlow, Jr. 1976. Crustacean larval behavior as an indicator of sublethal effects of an insect juvenile hormone mimic. In: Estuarine processes, vol. I: uses, stresses, and adaptation to the estuary. Academic Press, NY. pp. 279-289.

Forward, R.B., Jr., and J.D. Costlow, Jr. 1978. Sublethal effects of insect growth regulators upon crab larval behavior. Water, Air, and Soil Pollution 9:227-238.

Klein, M.L., and J.L. Lincer. 1974. Behavioral effects of dieldrin upon the fiddler crab, *Uca pugilator*. In: Pollution and physiology of marine organisms. F.J. Vernberg and W.B. Vernberg. Academic Press, NY. pp. 181-196.

Koenig, C.C. 1977. The effects of DDT and mirex alone and in combination on the reproduction of a salt marsh cyprinodont fish, *Adinia xenica*. In: Physiological responses of marine biota to pollutants. F.J. Vernberg, A. Calabrese, F.P. Thurberg, and W.B. Vernberg, eds. Academic Press, NY. pp. 357-376.

Livingston, R.J., N.P. Thompson, and D.A. Meeter. 1978. Long-term variation of organochlorine residues and assemblages of epibenthic organisms in a shallow north Florida (USA) estuary. Mar. Biol. 46:355-372.

Livingston, R.J., R.L. Iverson, R.H. Estabrook, V.E. Keys, and J. Taylor, Jr. 1974. Major features of the Apalachicola Bay System: physiography, biota and resource management. Fla. Scient. 37:245-271.

Neff, J.M., and C.S. Giam. 1977. Effects of Aroclor® 1016 and Halowax® 1099 on juvenile horseshoe crabs *Limulus polyphemus*. In: Physiological responses of marine biota to pollutants. F.J. Vernberg, A. Calabrese, F.P. Thurberg, and W.B. Vernberg, eds. Academic Press, NY. pp. 21-35.

Nimmo, D.R., and L.H. Bahner. 1974. Some physiological consequences of polychlorinated biphenyl and salinity stress in Penaeid shrimp. In: Pollution and physiology of marine organisms. F.J. Vernberg and W.B. Vernberg. Academic Press, NY. pp. 427-444.

Nimmo, D.R., and R.R. Blackman, A.J. Wilson, Jr., and J. Forester. 1971a. Toxicity and distribution of Aroclor® 1254 in the pink shrimp, *Penaeus duorarum*. Mar. Biol. 11:191-197.

Nimmo et al., 1971b. Polychlorinated biphenyl adsorbed from sediments by fiddler crabs and pink shrimp. Nature (London) 231:50-52.

Odum, W.E., G.M. Woodwell, and C.F. Wurster. 1969. DDT residues absorbed from organic detritus by fiddler crabs. Science 1964:576-577.

Payen, G.G., and J.D. Costlow. 1977. Effects of a juvenile hormone mimic on male and female gametogenesis of the mud crab, *Rhithropanopeus harrisii* (Gould) (Brachyura: Xanthidae). Biol. Bull. 152:199-208.

Peakall, D.B. 1975. PCBs and their environmental effects. CRC Crit. Revs. Environ. Contr. 5:469-508.

Poole, R.L., and M. Willis. 1970. Effects of some pesticides on larvae of the market crab, *Cancer magister*, and the red crab, *Cancer productus*, and a bioassay of industrial wastes with crab larvae. Manuscr. Rep. Calif. Dep. Fish Game Mar. Resour. Reg. 70-15: 1-19.

Roesijadi, G., S.R. Petrocelli, J.W. Anderson, C.S. Giam, and G.E. Neff. 1976. Toxicity of polychlorinated biphenyls (Aroclor® 1254) to adult, juvenile and larval stages of the shrimp *Palaemonetes pugio*. Bull. Environ. Contam. Toxicol. 15:297-304.

Tester, P.A., and J.D. Costlow, Jr. 1979. The effect of the insect growth regulator Dimilin® (TH 6040) on the fecundity and egg viability of the marine copepod *Acartia tonsa* Dana (Copepoda: Calanoida). Submitted to Mar. Ecol.

Vernberg, W.B., P.J. DeCoursey, and J. O'Hara. 1977. Multiple environmental factor effects on physiology and behavior of the fiddler crab, *Uca pugilator*. In: Pollution and physiology of marine organisms, F.J. Vernberg and W.B. Vernberg. Academic Press, NY. pp. 381-426.

Walsh, G.E. 1972. Insecticides, herbicides, and polychlorinated biphenyls in estuaries. J. Wash. Acad. Sci. 62: 122-139.

Williams, A.B., and T.W. Duke. 1979. Chapter VI. Crabs (Arthropoda, Crustacea, Decapoda, Brachyura). In: Pollution ecology of estuarine invertebrates, C.W. Hart, Jr., and S.L.H. Fuller. Academic Press, NY. pp. 171-233.

IMPACT OF OIL ON THE COASTAL ENVIRONMENT

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It is generally recognized that petroleum and the activities that accompany the petroleum sector are having a major impact on many portions of the world's coastal environments. Certainly a similar impact is felt by noncoastal environments, but it appears that a number of factors combine to focus the major impact on coastal zones. For example, large amounts of petroleum are produced in coastal areas and even larger amounts are transported on coastal waters. These activities in an environment that is sensitive, yet harsh and unforgiving of mistakes, have made management of the petroleum sector difficult. Petroleum, unlike other pollutants such as radioactivity, heavy metals, and pesticides, can be seen, felt, and smelled. This fact and the massive releases in some oil pollution incidents have further compounded the problem in coastal environments. The sheer magnitude of the activities of the petroleum sector and the complex chemical and biological properties of petroleum have brought changes to affected portions of the coastal environment far greater than to non-petroleum related areas. Petroleum may be thought of as a driving force for many environmental modifications.

Exploration, drilling, and production of oil in the coastal environment have brought changes in the physical environment. Development of a major oil or gas field may involve extensive on-shore activities such as shipyard, fabrication plant, harbor, tank-farm, and pipeline construction. These projects may involve dredging and filling of wetlands, alternate uses of coastal waters, competition with established labor markets, and other social impacts. The coastal environment may be greatly modified by the petroleum sector before a barrel of oil is produced, transported, or refined. This modification of the physical environment may be viewed as a competition for the limited resources of the coastal zone. It can have far-reaching impacts on the natural ecosystem.

Offshore production and transportation of petroleum, river runoff, and municipal and industrial wastes combine to inject millions of tons of petroleum and petroleum products into the marine environment annually (Table 1) (National Academy of Science, 1975). The potential of these releases for damage to organisms and ecosystems has been the central theme for most of the petroleum-related environmental science studies for the past decade. The studies have been difficult for two major reasons: petroleum is a complex substance not readily characterized, and effects of petroleum on biota are highly variable and poorly understood. Since 1970 our knowledge of baseline levels, transport mechanisms, weathering, and biological effects of petroleum in the marine environment has increased dramatically. The balance of this report describes some of the planned and unplanned studies of oil pollution.

PETROLEUM AS A NATURAL SUBSTANCE

Composition

It is somewhat ironic that the coastal environment, the area most impacted by the petroleum sector, is the environment in which most petroleum originated in earlier

Table 1. Comparison of Estimates for Petroleum Hydrocarbons Annually Entering the Ocean, Circa 1969-1971

	Authority (millions of tons per annum)		
Source	MIT SCEP report (1970)	USCG impact statement (1973)	NAS workshop (1973)
Marine transportation	1.13	1.72	2.133
Offshore oil production	0.20	0.12	0.08
Coastal oil refineries	0.30	—	0.2
Industrial waste	—	1.98	0.3
Municipal waste	0.45	—	0.3
Urban runoff	—	—	0.3
River runoff*	—	—	1.6
Subtotal	2.08	3.82	4.913
Natural seeps*	?	?	0.6
Atmospheric rainout	9.0†	?	0.6
Total	11.08	?	6.113

***PHC input from recreational boating assumed to be incorporated in the river runoff value.**

†Based upon assumed 10 percent return from the atmosphere.

Source: Reproduced from *Petroleum in the Marine Environment*, 1975, with the permission of the National Academy of Sciences, Washington DC.

periods of geological time. Petroleum consists of the remains of the microscopic plants and animals that lived in coastal seas. During the millions of years since this once-living material was deposited in sediment, the biogenic components have undergone chemical reactions that have produced the complex mixture we call petroleum. The compositions of petroleums vary greatly with source, but the qualitative composition is surprisingly similar. These common compositional features are thought to reflect the common origin and similar chemical history of petroleum. In every sense, then, petroleum is a natural substance. But it is a natural substance that has been buried for millions of years, during which time it has generated chemical components that do not exist in the normal food web of ecosystems. Since some of the compounds produced during petroleum generation are toxic to biota, it is appropriate to consider briefly and generally the composition of petroleum.

Petroleum is composed of compounds of carbon and hydrogen (hydrocarbons) and minor amounts of organic molecules that also contain nitrogen, sulfur, or oxygen (NSO compounds). The hydrocarbons present in petroleum belong to five classes:

- normal alkanes
- branched chain alkanes
- cycloalkanes or naphthenes
- aromatics
- naphtheno-aromatics

The NSO compounds are diverse in structure, not easily classified but important from an environmental viewpoint based on their toxic properties. Structural formulas of typical compounds from each hydrocarbon class plus NSO compounds are shown in Figure 1. Petroleums are characterized by the relative amounts of these molecular types that they contain and by the molecular weight distribution of each type. The molecular weight distribution as reflected in boiling range is the property that defines the various fractions of petroleum produced by a refinery. The boiling range fractions such as gasoline, kerosene, and fuel oil are those most well known. Despite the fact that millions of barrels of crude oil are used every day, most people have not had direct experience with it. Figure 2 graphically illustrates the molecular

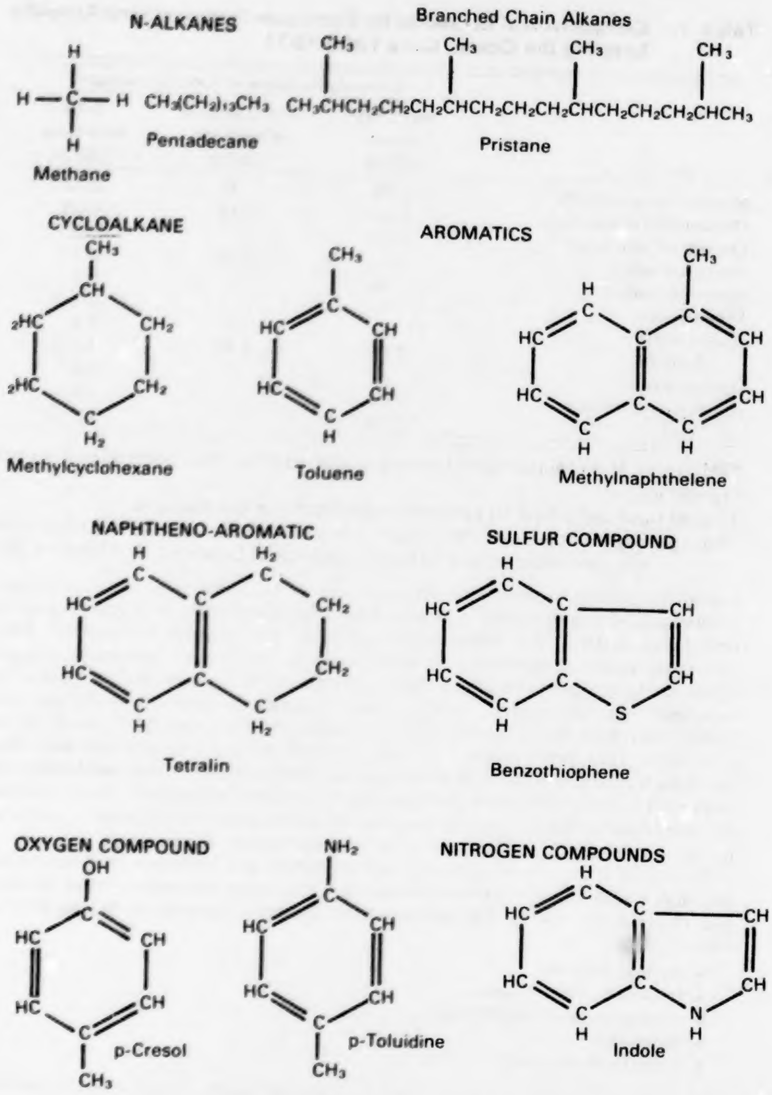
N-ALKANES
Branched Chain Alkanes
Methane
CH3(CH2)13CH3
Pentadecane
CH3CHCH2CH2CH2CHCH2CH2CH2CHCH2CH2CH2CHCH3
Pristane
CYCLOALKANE
Methylcyclohexane
AROMATICS
Toluene
Methylnaphthelene
NAPHTHENO-AROMATIC
Tetralin
SULFUR COMPOUND
Benzothiophene
OXYGEN COMPOUND
p-Cresol
p-Toluidine
NITROGEN COMPOUNDS
Indole

Figure 1. Classes of compounds found in petroleum.

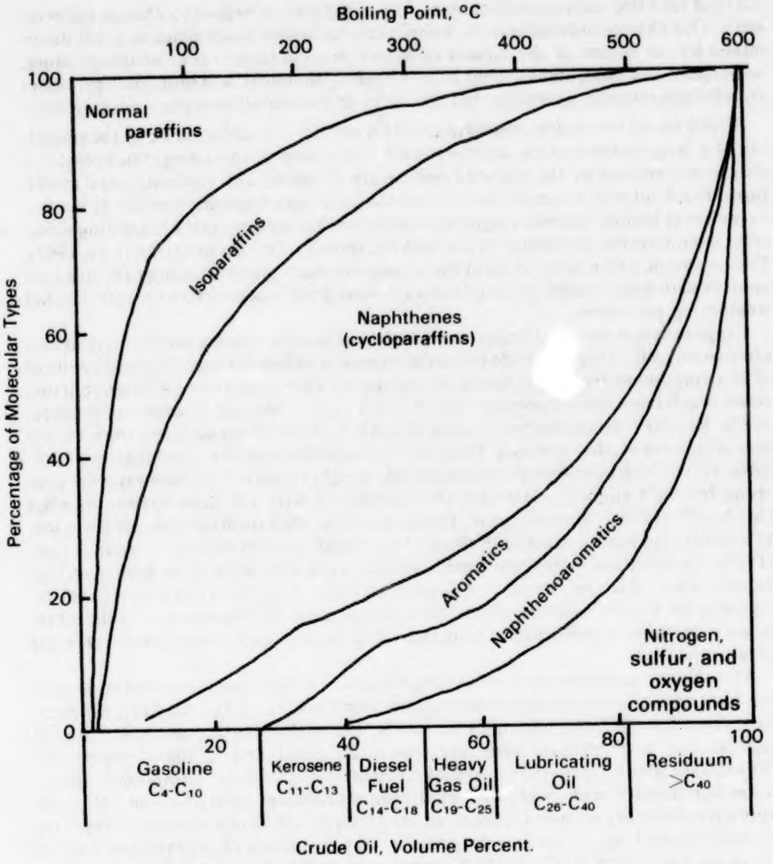

Figure 2. Distribution of the chemical compounds of a crude oil. (From John M. Hunt, *Petroleum Geochemistry and Geology.* W. H. Freeman and Company, Copyright© 1979.)

composition of the major boiling range materials found in a typical crude oil (Hunt, 1979). Data contained in this figure are useful for predicting the fate or weathering pattern of a petroleum or refined product. For example, a spill of gasoline with a low boiling range evaporates in the marine environment. Although different crudes may have compositional curves very different from those in Figure 2, general conclusions as to the potential toxicity of an oil or oil product can be made. For example, mid-boiling aromatics that are regarded as relatively toxic are present in high concentrations in kerosene and diesel fuel. Nitrogen, sulfur, and oxygen compounds are also toxic, but those in the high-boiling range (greater than 500° C) may be molecules that are too large to have biological activity. On the other hand, NSO compounds in the diesel fuel range are low in concentration, but they are small, water soluble, and biologically active, and thus are a cause for environmental concern.

Weathering

The potential for biological and aesthetic damage by chronic or acute petroleum release is a function of the weathering profile of the oil. When an oil is spilled or discharged into the marine environment, the composition begins to change immediately. This change in composition, known as *weathering*, takes place at a rate determined by the nature of the oil and by environmental factors such as temperature, wind speed, sea state, and nutrient level. Physical, chemical, and biological processes are all continuously operative, but the rates of individual processes vary greatly.

A spill on marine waters results in rapid formation of a surface slick by the spreading of a large portion of the discharged oil. The extent of spreading (thickness) of a slick is determined by the quantity and nature of the oil and environmental conditions. Fresh oil with a compositon such as that in Figure 1 spreads rapidly. However, as the lower boiling fraction evaporates, attractive forces among the remaining molecules overcome the spreading forces, and the spilled oil ceases to spread (Fay, 1969). Formation of a thin layer of oil at the air-sea interface provides a large surface area contact with both atmosphere and seawater that greatly accelerates the rate of other weathering processes.

Evaporation is the most important weathering process during the first day or two after an oil spill. A typical crude oil can be expected to lose between 25 and 50 percent of its components from a surface slick owing to evaporation alone. A 75 percent loss could result from the evaporation of a No. 2 fuel oil (National Academy of Science, 1975). Results from numerous studies indicate hydrocarbons with less than 15 carbon atoms are readily removed from oil by evaporation under usual marine conditions. Of the hydrocarbons in this molecular weight range ($<C_{15}$), saturates are generally lost at a slightly faster rate than aromatics with the same carbon number (McAuliffe, 1976a). Harrison et al. found, however, that isopropyl benzene was lost at a faster rate than *n*-nonane in three of five small spills of South Louisiana crude (1975). Evaporation rates from freshly spilled crude oils, while quite high, decrease exponentially. Loss of the most volatile fraction ($<C_{10}$) results in a significant increase in the density and viscosity of the remaining oil. As the viscosity of the oil increases, molecular diffusion within the oil decreases and evaporation rates are greatly reduced.

Most petroleum components that are aromatic or contain nitrogen, sulfur, or oxygen have seawater solubilities in the milligrams per liter range (ppm). Alkyl benzenes have solubilities of 50 to 500 mg/l (McAuliffe, 1966) while the seawater solubilities of naphthalene and phenanthrene are about 22 mg/l and 1 mg/l respectively (Eganhouse and Calder, 1976). Actual concentrations of these compounds in seawater equilibrated with crude or fuel oils (an oil to water ratio of about 1:9) never approach solubility values. Concentrations of naphthalene in seawater are reported to be less than 1 mg/l owing to the preferential partitioning of naphthalene into the oil phase (Anderson et al., 1974). Comparisons of the rates of loss by evaporation and solution have been made in several studies. Harrison et al. (1975) reported that the rate of evaporation of isopropyl benzene was about two orders of magnitude faster than the rate of solution. Cyclohexane and benzene have similar vapor pressures, but benzene has a much higher solubility. McAuliffe (1976a) found that the two compounds were lost from crude oil slicks at similar rates, showing that loss by solution was minor.

Emulsification is an extremely important process in many spills. An emulsion is the colloidal suspension of a liquid within a second immiscible liquid. Water-in-oil emulsions are formed by the incorporation of tiny droplets of seawater within an oil. Water-in-oil emulsions often result in formation of a stable gel, brownish in color, that is referred to as chocolate mousse or simply mousse. The seawater content of a mousse is generally between 50 and 80 percent. The tendency of an oil toward mousse formation appears to be largely a function of he concentration of surface active compounds present in the oil. Asphaltenes and NSO compounds have been suggested as the probable emulsifying agents (MacKay et al., 1973). Mousse formation generally results in a decrease in the rate of weathering of an oil. The greater density, viscosity, and volume of the mousse results in a thickening of the layer of oil and a decrease in the surface area/volume ratio. Mousse formed from Kuwait crude was exposed for 2 years in floating enclosures without effective degradation (Davis and Gibbs, 1975). An oil-in-water emulsion or dispersion results from the suspension of small particles/droplets of oil within the water column. Oil is physically forced into the water column by turbulence at the surface, generally wave action. The degree of turbulence determines the quantity and depth of mixing while oil density and particle size determine residence time in the water column (Forrester, 1971).

Sedimentation is a process that determines the fate of some heavy oils in the marine environment. In the case of a few spills involving Bunker C fuel oil, a significant portion of the oil quickly dispersed within the water column in the form of droplets or globules that had a specific gravity equal to or greater than the surface seawater (Conover, 1971). These particles can sink in the water column and be moved about by subsurface currents (Conomos, 1975). Less dense crude and fuel oil droplets dispersed in surface waters can be sedimented by adsorption to suspended mineral particles. This process could be particularly important in more turbid coastal waters. Oil droplets may also be ingested by zooplankton and excreted in fecal pellets that have density greater than seawater (Conover, 1971).

Photochemical oxidation is probably the most important of the various abiotic chemical reactions that occur during the weathering process. Ultraviolet radiation is absorbed by selected molecules, especially aromatics, that react with oxygen to form oxygenated intermediates such as hydroperoxides. These oxygenated intermediates are generally much more soluble and toxic than the parent compound (Larson et al., 1976). Products that result from irradiation of oil include organic acids, alcohols, esters, aldehydes, ketones, phenols, and sulfoxides (Parker et al., 1971). Photochemical and abiotic oxidation of crude oil during the weathering process may be largely responsible for the high NSO and asphaltene content of some tarballs and tarry residues that are difficult to account for by simple evaporation and solution losses (Frankenfeld, 1973).

Biodegradation is an important process in the further weathering of oil that has dispersed or dissolved in seawater. Although microbial degradation plays a far more significant role in weathering, macroorganisms also degrade oil. Marine animals of all sizes and genera may purposefully or accidentally ingest oil particles during feeding. A portion of the oil is absorbed and incorporated into tissues while the bulk is eliminated in the feces (Conover, 1971). Petroleum hydrocarbons are depurated from tissues at various rates depending upon the type of tissue and organism.

A broad spectrum of mcroorganisms is active in the metabolism of petroleum that enters the marine environment. Cyanobacteria (blue-green algae) and microalgae (Cerniglia et al., 1980), bacteria (Zobell, 1969), fungi (Walker et al., 1973), and yeasts (Klug and Markovetz, 1967) have been reported capable of degrading oil or oil components. Hydrocarbon-utilizing microorganisms are ubiquitous in the oceans. However, their concentrations vary, with highest counts in areas of chronic petroleum pollution (Walker and Colwell, 1976). Individual species differ in ability to grow on petroleum hydrocarbons as a sole carbon source. Often a species may be able to utilize a given aromatic ring structure while unable to grow on others. Maximum degradation of petroleum is accomplished by the combined and sequential attack of a mixed population (Horowitz et al., 1975).

Numerous studies indicate preferential utilization of *n*-alkanes during early stages of microbial degradation of crude oils (Jobson et al., 1972). Isoprenoid hydrocarbons such as pristane and phytane are often degraded at a slower rate than *n*-alkanes (Westlake et al., 1974). Walker et al. (1976) report alkanes degraded to a greater extent than cycloalkanes in two crude oils. A wide variety of aromatic and naphtheno-aromatic compounds may also be utilized by microorganisms. A larger percentage of aromatics were degraded than saturates in the study of Walker et al. (1976).

Microbial degradation rates in the ocean may become nutrient-limited. Nitrogen (NO_3^-, NH_4^+) and/or phosphorus (PO_4^{-3}) concentrations can determine degradation rates especially in offshore areas.

All of these physical, chemical, and biological processes act to keep the level of petroleum low in the environment. It is fair to state that they work effectively. Many large oil spills are dispersed before they become a biological problem. For example, the *Argo Merchant* spill off Cape Cod and the Ekofisk blow-out did not lead to heavy accumulations. The massive Ixtoc I spill did lead to accumulations over a 960-km (600-mi) long track; however, foul weather and perhaps biological activity finally removed most of the oil from the Texas coast. Nevertheless, the potential for damage to marine plants and animals by chronic and acute inputs of petroleum is a serious problem. Extensive studies have been done to help define and resolve this problem.

BIOLOGICAL EFFECTS

The impact of petroleum on marine plants and animals is of central importance for scientists and regulatory agencies concerned with the petroleum sector. This problem is a real challenge to scientists because it has many levels of expression, all of which are important and perhaps interrelated. Determination of the acute toxicity of petroleum and petroleum products toward marine plants and animals is complicated by the fact that the sensitivity of various marine organisms is highly variable. Generalizations are difficult to make. It is somewhat the case that levels of petroleum in the environment high enough to cause acute toxicity are also high enough to trigger cleanup efforts. As a consequence, much of the research on the biological effects of oil during the past decade has been on the sublethal effects of oil. When one considers that a given petroleum may contain more than 300 compounds, many of which may be deleterious to some biota, the magnitude of the problem is clear.

Research programs dealing with the harmful effects of petroleum form a matrix. One side of the matrix is the level of organization of the biota. This ranges from impact on key metabolic reactions such as photosynthesis and nitrogen fixation to effects on bacteria, phytoplankton, microzooplankton, zooplankton, larval animals, invertebrates and large fish and finally to whole communities or ecosystems. The other side of the matrix is individual petroleum compounds, various fuel oils produced by the petroleum industry, and crude oils. The scientific community is divided as to which matrix members are environmentally most critical. Some argue to focus on the effects of oil on key processes such as photosynthesis or respiration, others point to the need to understand the impact on commercial fish and shellfish, while others contend that only the total ecosystem should be studied. The variety of life and the chemically complex nature of petroleum determines that only limited studies can be made of this matrix with the human and financial resources that are available. Proceedings of the National Academy of Sciences workshop on marine environmental quality provided useful guidelines for biological studies (National Academy of Sciences, 1971). This document pointed out the inadequacy of acute toxicity tests. It stated that the greatest scientific benefit would accrue if "intensive research is focused on relatively few kinds of organisms and systems." Suggested criteria for system selection include available basic data, economic importance, magnitude, and exemplary nature. The decade of the 1970s has brought forth studies along these lines ranging from microbes to ecosystems.

How toxic is petroleum to marine organisms? The answer to this question is vital to an assessment of environmental damage associated with specific petroleum pollution incidents that occur on a daily basis in the coastal zone. The answer is also vital to assessment of the hazards of long-term chronic pollution of harbors, bays, and estuaries. Biological effects observed when experimental organisms are exposed to water-soluble fractions of oil, oil dispersed in seawater (accommodated oil), or specific petroleum components have received detailed study. Mortality of birds and intertidal animals as a direct result of oil coating may be described as a form of acute toxicity but will not be discussed here.

Our fundamental understanding of the nature of petroleum toxicity has developed from laboratory studies of acute toxicity. A generally accepted method of expressing acute toxicity of a given pollutant is the 96-hr LC_{50}. The LC_{50} value of a pollutant is the concentration of the toxicant required to kill 50 percent of the test organisms within 96 hours. Three variables are immediately obvious in the determination of an LC_{50} for petroleum: (1) the oil tested, (2) the form of oil presented (water-soluble fraction or dispersion), and (3) the organism and life stage tested. Each of these variables affect the LC_{50} value obtained and thus partially explains the wide range of values reported in the literature. Table 2 presents a compilation of LC_{50} data largely obtained with water-soluble fractions (WSF) of oils or pure compounds. The data of Table 2 indicate fuel oils are generally more toxic than crude oils to a wide variety of organisms. Fuel oils (No. 2 and No. 6) contain larger percentages of toxic naphthalenes and phenanthrenes than do crude oils. Rossi et al. (1976) found two fuel oils more toxic to the polychaete *Neanthes arenaceodentata* than two crude fuel oils. The higher toxicity of the fuel oils was attributed to their higher content of naphthalenes. Tissues from animals exposed to WSF from No. 2 fuel oil contained a higher concentration of naphthalenes than animals exposed to WSF from crude oil. Crude oil-exposed animals had higher tissue levels of alkyl benzenes. Accumulation of naphthalenes in organs of *Fundulus similus* exposed to WSF of No. 2 fuel oil has been documented by Dixit and Anderson (1977). Concentrations of total naphthalenes in excess of 200 ppm in the brain were found to correlate with loss of locomotor and regulatory capabilities. Affected fish placed in clean water returned to normal swimming behavior within 3.5 hours, and brain levels of naphthalenes were found to be about 200 ppm. Winters and Parker (1977) have reported that WSF of fuel oils also contain higher concentrations of phenols, anilines, indoles, and quinolines than do crude oils. Some of these one- and two-ring aromatic compounds containing oxygen or nitrogen may be formed during catalytic cracking and reforming of crude oil in refinery processes. The differences in toxicities of No. 2 fuel oils to microalgae have been demonstrated to depend upon differences in concentration of these minor components. Winters et al. (1976) tested the toxicity of WSF from four No. 2 fuel oils on the growth of microalgae. Water-soluble fractions from two of the oils were lethal to the two blue-green algae tested. The observed toxicity to blue-greens was due to a higher concentration of anilines in these oils. Para-toluidine, a methyl aniline, was found to be toxic to *Agmenellum quadruplicatum* at a concentration of 100 μg/l. Similarly in another study, perinaphthenone was found to be largely responsible for the high toxicity of WSF of a No. 2 fuel oil to green algae (Winters et al., 1977). Perinaphthenone, a three-ring aromatic ketone, was present at a concentration of 200 μg/l in the WSF. Toxicity of the pure compound alone was demonstrated at a concentration of 250 μg/l. Crude oils also exhibit a wide range of toxicities in bioassays, no doubt owing to their diverse chemical and physical properties.

Oil has generally been presented to test organisms as an oil-in-water dispersion (OWD) or as a water-soluble fraction prepared from the oil. Neither method produces a stable concentration of oil in seawater. The stability of an OWD depends upon the size of droplets, specific gravity of the oil, and seawater circulation within the test chamber. Evaporation of volatile components quickly decreases the concentration of WSF even in nonaerated aquaria (Winters and Parker, 1977). Maintenance of constant concentrations of oil in seawater during exposure is greatly facilitated by use of flow-through test chambers. Flow-through systems provide additional advantages such as maintaining dissolved oxygen concentration with less or no aeration, minimizing problems associated with bacterial contamination, and removing waste products from test animals.

The toxicities of individual compounds known to occur in petroleum have also been tested (Neff et al., 1976). Generally, toxicity of naphthalenes was greater than that of benzenes and increased with degree of alkyl substitution of a given aromatic ring system (Table 2). Neff et al. (1976) found phenanthrenes more toxic than naphthalenes to a species of polychaete worm.

Table 2. Comparison of 96-Hr LC_{50} Values (ppm) for Marine Animals Tested with Various Oils or Aromatic Hydrocarbons

Marine animal	Crude oils	Fuel oil	Benzenes	Naphthalenes
Polychaete	9.5 - 12.5 (Neff et al., 1976)*	2.0 - 8.4 (Rossi and Anderson, 1976)		2.0 - 3.8 (Neff et al.,1976)
Amphipod	0.6 - >2.4 (Linden, 1976) (Lee et al., 1977)	0.2 - 1.0 (Linden, 1976) (Lee et al., 1977)		2.7 (Lee and Nicol, 1978)
Shrimp	6.0 - >19.8 (Neff et al., 1976)	1.0 - 6.6 (Neff et al., 1976)	1.0 - 27.0 (Neff et al.,1976) (Benille and Korn, 1977)	0.7 - 2.4 (Neff et al., 1976)
Fish	5.5 - >19.8) (Anderson et al., 1974)	3.9 -6.3 (Anderson et al., 1974)	3.9 - 12 (Benille and Korn, 1977)	
Crustacean larvae	0.8 - 4.9 (Mecklenburg et al., 1977) (Wells and Sprague, 1976)	1.0 - 1.5 (Neff et al.,1976)	2.1 - 108 (Caldwell et al., 1977)	0.6 - >2.0 (Caldwell et al., 1977)

*Indicates reference.

Bioassays to determine acute toxicity of petroleum to a species should be conducted with the most sensitive stage in the life cycle of that species. Allen (1971) tested the effects of WSF from 16 crude and fuel oils on the development of sea urchin eggs. Little or no effect was seen on fertilization; however, 11 oils affected cleavage at concentrations of 6 percent WSF or higher. Nicol et al. (1977) reported that similar concentration of WSF (6 percent) from a No. 2 fuel oil also affected cleavage of eggs and larval development in sand dollars. Larvae and juvenile forms have often been found more vulnerable to oil than are adults (Table 2). Wells and Katz suggested that the greater sensitivity of crustacean larvae was linked to molting (Wells, 1972). Subsequent work by Mecklenburg et al. (1977) indicated molting shrimp larvae were about five times more sensitive than nonmolting larvae. Recent studies report, however, that early life stages are not the most sensitive for some species of shrimp (Neff et al., 1976) and polychaete (Rossi and Anderson, 1976).

Numerous studies have reported sublethal effects of petroleum on marine organisms. A sublethal effect is any abnormal response observed during exposure of an organism to nonlethal concentrations of toxicant. Sublethal studies are valuable in the assessment of the potential for damage at oil concentrations that are more probable in the environment than LC_{50} values. Metabolic rate measurements such as respiration rates generally show an increase in metabolism owing to stress at low oil concentrations (Hargrave and Newcombe, 1973). Respiration rates continue to increase with increased oil concentrations until near-lethal concentrations result in reduced rates. Bradycardia (slowing of the heart) was measured in sea catfish by Wang and Nicol at 0.01 ppm (less than 10 percent of the LC_{50}) of a No. 2 fuel oil (Nicol and Wang, 1977). Feeding responses deteriorated at about 0.038 ppm (27 percent of the LC_{50}). Atema and Stein (1974) reported feeding behavior of the lobster was affected by relatively low concentrations of crude oil. Recently, modifications to the feeding behavior of marine copepods have been reported at concentrations of 0.25 ppm of fuel oil accommodated in seawater (Berman and Heinle, 1980). Total suppression of feeding or suppression of feeding on small particles between 7 and 15 μm in diameter was observed. Feeding on particles larger than 15 μm was increased in some cases.

Chemotaxis of a snail (Jacobson and Boylan, 1973) and phototaxis of barnacle larvae (Donahue et al., 1977) were altered by WSF from fuel oils.

Many of the biological effects attributed to stress from sublethal concentrations of pollutants should be reflected in growth rate. The growth rate of an organism is an ecologically significant parameter that can easily be measured in the laboratory. Reduced growth rate of the polychaete worm *Neanthes arenaceodentata* has been reported by Anderson in 3 percent WSF of a No. 2 fuel oil (Anderson,1977). The concentration of total hydrocarbons was 180 ppb; total naphthalene concentration was 60 ppb. Neff et al. (1976) exposed larvae of the mud crab *Rhithropanopeus harrisii* to WSF of No. 2 fuel oil for 6 months. Survival of larvae in the lowest two concentrations (0.16, 0.31 ppm total hydrocarbons) was similar to controls (~ 90 percent). Survival in 0.63, 0.94, and 1.26 ppm concentrations were 76, 30, and 6 percent, respectively. After 6 months, the mean size of crabs in the control group was larger than that of the 0.16, 0.31, and 0.63 ppm groups but smaller than that of the 0.94 and 1.26 ppm groups. Larvae of the mud crab *Eurypanopeus depressus* (Smith) were exposed to 4.3 and 8.7 ppm concentrations of WSF of Kuwait crude and development followed through crab stage five (Cucci and Epifanio, 1979). Increased mortality and duration of intermolt periods were observed in larvae exposed continuously from hatching. Larvae not exposed to WSF before zoea stage three showed slower growth than controls but no higher mortality. An extra morphologically abnormal megalopa stage was observed for some exposed individuals. The percentage of animals that exhibited the extra abnormal stage increased in the higher concentration of WSF. Decreased growth of larvae of the amphipod *Gammarus oceanicus* during a 60-day exposure to WSF from a Venezuelan crude was reported by Linden (1976).

A few studies have investigated the uptake of petroleum hydrocarbons presented to organisms in forms other than oil-in-water dispersions or water-soluble fractions. Bioaccumulation of fuel oil hydrocarbons adsorbed onto kaolin particles and ingested by *Mytilus edulis* has been reported by Fossato and Canzonier (1976). Tissue concentrations were measured in excess of 1,000 times the exposure concentration. No accumulation of hydrocarbons was observed in the polychaete *Neanthes arenaceodentata* held for 28 days in sediments contaminated with No. 2 fuel oil. Also, ^{14}C-2-methylnaphthalene spiked detritus was fed to the animal for 16 days without accumulation of radioactivity in tissues (Rossi, 1977). A detritivorous clam, *Macoma inquinata* was found to take up naphthalenes from seawater but not sand or detritus contaminated with Prudhoe Bay crude (Roesijadi et al., 1978).

Concentrations of petroleum hydrocarbons that build up in tissues of organisms during exposure to oil decrease rapidly upon return to clean seawater. The rate and extent of depuration have been determined in a number of studies. Many of the studies have utilized molluscs as test organisms (Lee et al., 1972). One of the most detailed studies of accumulation and depuration has recently been described by Clement et al. (1980). The clam *Macoma balthica* was found to fractionate hydrocarbons present in a dispersion of Prudhoe Bay crude by preferential retention and release of certain compound classes and homologs. Depuration was followed for 60 days, at which time exposed animals still contained tissue hydrocarbon concentrations of about 110 μg/l (wet weight) as compared with about 14 μg/l for controls. Fossato and Canzonier (1976) suggested that rapid loss of petroleum hydrocarbons observed from some *Mytilus edulis* was associated with spawning. Similar rapid loss of polychlorinated biphenyls from oysters during spawning has been proposed by Lowe et al. (1972). Rossi and Anderson exposed male and gravid female polychaetes, *Neanthes arenaceodentata*, to WSF of No. 2 fuel oil for 24 hours. The animals were then transferred to clean seawater. Male worms were found to depurate naphthalenes to undetectable levels within 17 days. Gravid females showed only minor release of naphthalenes during this period. Females analyzed within 24 hours after spawning contained barely detectable levels of naphthalenes. Zygotes and trochophore larvae produced by the spawn had naphthalene concentrations similar to those of the gravid females prior to spawning (Rossi and Anderson, 1977).

Two major, highly innovative research programs to study the long-term effects of sublethal levels of petroleum and other pollutants at the ecosystem level were designed and tested in the seventies. Both of these made use of large microcosms that would simulate a natural ecosystem.

The Controlled Ecosystem Pollution Experiment (CEPEX) was intended to simulate the plankton community of open ocean waters (Menzel and Case, 1977). The CEPEX structures were flexible plastic columns enclosing about 500 m^3 (650 yd^3) of water. The CEPEX program used copper as the pollutant in most of their experiments, although a few oils were tested. A major conclusion of the program was that population structure and succession patterns of plankton are more useful as stress measurements than metabolic ones. The CEPEX concept could be applied to studies of oil impacts on pelagic plankton. Such studies would have been especially useful to test the impact of Ixtoc I oil in the vicinity of the well-head. Arnold showed that slightly weathered Ixtoc oil was acutely toxic to the eggs and newly hatched larvae of redfish (Arnold et al., 1979). Canadian workers have conducted long-term CEPEX type experiments by adding petroleum to plastic cylinders in freshwater lakes (Scott and Shindler, 1978). They reported that the bacteria populations were enhanced. However, the zooplankton populations in the oiled ponds were drastically reduced when the ice cover on the pond melted. The phytoplankton community structure was changed by the oil but no clear pattern was obvious. The experiments did not include a benthic element.

Large microcosms established at the Marine Ecosystem Research Laboratory (MERL) have both a pelagic (water column) and a benthic (sediment) component. This allows studies of the transport of pollutants such as oil to the sediment and of the effects on plankton and benthic animals such as small bivalves and polychaete worms. The MERL microcosms are fiberglass tanks 5.5 × 1.8 m (18 × 6 ft) in diameter with 40 cm (16 in) of sediment in the bottom. They have been shown to simulate Narragansett Bay to an extent that pollution experiments are realistic when a control tank is used (Pilson et al., 1979). Numerous experiments have been done in this system using No. 2 fuel oil as the pollutant. It was found that added oil was 40 to 60 percent removed by evaporation, but significant quantities of both saturated and aromatic hydrocarbons were transported to the bottom by adsorption on sinking particles (Gearing et al., 1979). At water column concentrations of 93 ppb total hydrocarbon, most species of benthic animals declined relative to a control tank. These effects were obvious for a year (Grassle et al., 1980). Microcosm experiments are difficult because they involve large facilities and team research, but they are very promising as a tool for evaluating the potential effects of petroleum and other toxic substances on the marine environment.

Planned and unplanned spills of oil in salt marshes have been studied (Lytle, 1975). These opportunistic studies have not provided in-depth multidisciplinary data. They have shown that more toxic oils such as No. 2 fuel oil can have long-term impacts on marsh plants. Small spills of crude oil have an impact related to the amount of oil spilled and the care exercised in the cleanup operations. The study of the Miguasha Marsh, Quebec, following a spill of Bunker C fuel oil, showed that oil could persist in sediment and be redistributed by tidal waters. Nevertheless, from a revegetation point of view, manual cleanup without burning was recommended by Vandermeulen and Ross (1977).

PETROLEUM IN THE ENVIRONMENT

During the seventies, our knowledge of the kind and concentration of petroleum in the environment has increased enormously. There are several reasons for this. The driving force is the increased transport and use of petroleum coupled with a proportionate concern for the impact of oil on the environment. During this time enabling developments included the rapid development of analytical instrumentation such as gas and liquid chromatography, gas chromatography coupled to mass spectrometry (GC/MS), and compact, dedicated computer-assisted data acquisition and analysis systems. All of these developments were possible because private, state, and federal agencies made substantial funds available. It may be argued that the limiting factor for environmental chemistry is the number of well-trained scientists with ideas.

Programs that have undertaken to measure the baseline level of petroleum hydrocarbons in the marine environment clearly reflect this growing analytical sophistication in the data and in the cost of obtaining the data. In 1971-1972 the U.S. International Decade of Ocean Exploration (IDOE) carried out baseline studies of pollutants including petroleum in the U.S. coastal waters. These investigations by a group of university scientists, despite being only one year in duration, stimulated a follow-up conference to recommend "that a continuing research program to determine inputs, dispersal paths and present levels of . . . petroleum hydrocarbons in representative plants and animals of coastal and open ocean zones be immediately initiated with the objectives of evaluating hazards to living processes and of defining sources of these materials (International Decade of Ocean Exploration, 1972). Such a continuing research program has in fact been operative during the decade although it is a patchwork of various organized programs and efforts by individual scientists.

Beginning in 1974, the U.S. Bureau of Land Management (BLM), initiated a comprehensive study of offshore regions, mostly outer continental shelf areas, of the United States that were being considered for petroleum exploration. Measurements of petroleum hydrocarbon baselines for water, biota, and sediment were major goals for this program. This program involved many of the marine organic geochemists in the United States. At this time no single document summarizes the overall results of this program although such a document might be useful. The results where published and appropriate are cited herein without identification of the funding source.

The detection of petroleum in the environment requires the same technical operations used by organic geochemists in investigations of the chemical fate of biogenic material in a geological setting, similar to the work of those using geochemical techniques to prospect for petroleum. Thus it is not surprising that some of the first and best studies of petroleum in the marine environment came from established geochemistry laboratories. The late Max Blumer recognized the problem of oil in the environment and applied his knowledge and experience to dealing with it. Blumer et al. (1970) made a detailed study of the fate of 4,400 barrels of No. 2 fuel oil that was lost in a spill in Buzzards Bay, Massachusetts. These workers called the attention of the scientific community to several general points:

- Based on their composition, fuel oils are especially hazardous.
- Fuel oils can penetrate sediment and remain for months and years.
- Organisms can take up and retain aromatic hydrocarbons.
- Routine gas chromatographic methods are suitable for marine pollution events.

These observations may seem casual now but in 1969 they had an impact.

Farrington and Quinn (1973) demonstrated that low levels of petroleum hydrocarbons are present in the sediment and clams of Narragansett Bay and that sewage effluents and small oil spills are probably sources. Similar patterns of hydrocarbons were reported for New York Bight sediments (Farrington and Tripp, 1977). This suggests that in some circumstances petroleum can build up in offshore sediments. MacLeod et al. (1976) found a gradient of petroleum hydrocarbon concentration at two sites in the Strait of Juan de Fuca which was related to known upstream seepage. The presence of arenes (aromatic hydrocarbons) was stated to be the strongest evidence. Gearing et al. (1976) attribute the levels of hydrocarbons detected in the northeast Gulf of Mexico sediments to natural sources except for sites located near the Mississippi River. In general *n*-alkanes are not good indicator molecules for petroleum pollution because they are also natural products. However, using the planktonic blue-green algae, *Trichodesmium* sp. with a known simple *n*-alkane pattern, Parker et al. (1972) were able to show that massive natural blooms of the algae had become associated with a suite of *n*-alkanes derived from petroleum (Table 3).

A 4-year comprehensive study of petroleum hydrocarbons in water, biota, and sediment from the south Texas shelf was made by Parker, Giam, Winters, and Scalan (Parker et al., 1976). Several hundred samples were analyzed, and, with only a few exceptions, the levels of petroleum derived hydrocarbons were so low as to be undetectable. Neuston and some zooplankton tows showed unresolved humps in their GC and the presence of aromatic hydrocarbons, both of which indicate petroleum. This pollution is due to micro-tarballs that at times were taken near the surface. The study reached its goal, which was to affirm that the baseline level was low in a virgin area.

It is known that pelagic tar (floating tarballs) is present in tanker routes and near harbors. Much of these data have been summarized by Butler, Morris, and Sass (Butler et al., 1973). The biological consequence of this steady input is not known. When the material collects heavily on beaches, it is known to be a minor economic problem for beach users and motel operators. McAuliffe points out what must be a general truth for petroleum pollution in many areas: "The quantity of hydrocarbon in the waters and recent sediments of the oceans is small compared to total additions, indicating that destructive mechanisms are operative" (McAuliffe, 1976b).

The use of bivalves as sentinel organisms for detecting levels of pollutants in U.S. coastal waters has been the strategy of the Mussel Watch program. These animals have been monitored for levels of four categories of marine pollutants: heavy metals, transuranic elements, halogenated hydrocarbons, and petroleum hydrocarbons. Beginning in 1976 a 3-year surveillance of hydrocarbon levels in mussels and oysters (in the Gulf of Mexico) was made. Early results from the year-one and -two collections showed that low levels of polynuclear aromatic hydrocarbons (PAH) were present in many samples from both coasts and that sites with elevated levels were

Table 3. *n*-Alkanes and Isoprenoids in *Trichodesmium* Sp. (Percent Composition)

	Location			
Carbon Number	#1 Off Port Aransas, Texas	#2 Off Bayou La Fousche, Louisiana	#3 Off Freeport, Texas	#4 Off Houma, Louisiana
15	Tr	.3	.2	.4
16	2	1.2	.9	1.0
Pristane	—	Tr	.9	1.4
17	95	94.1	30.5	6.0
Phytane	—	Tr	1.4	1.5
18	1	.4	2.2	2.1
19		.4	3.2	2.4
20		.2	3.8	2.6
21		.1	3.9	2.3
22		.3	4.2	3.0
23		.2	4.7	2.7
24		.1	4.3	2.7
25		.1	4.9	3.6
26		.1	5.6	4.3
27		.1	4.7	4.3
28		.1	4.6	4.7
29		.1	3.8	5.3
30		.1	2.6	6.6
31		.1	1.6	6.3
32		.1	1.8	7.3
33		.1	1.9	6.5
34		Tr	1.8	6.1
35		Tr	2.8	9.7
36		—	2.4	6.0

detectable (Goldberg et al., 1978). Petroleum and pyrolytic sources must both be considered to explain the variety of compounds reported. Based on these data and other data reported at the National Academy of Science (NAS) workshop, bivalves are promising indicators of chemical pollution. (See Table 4.)

During the decade of the seventies, a number of planning and design meetings were held to consider how best to study the petroleum pollution problem. Some of these were strongly directed at sampling and ecological problems (Goldberg, 1972). Some were directed at specific locations or areas (MacLeod et al., 1976), and some were forerunners of massive federal programs (Parker, 1974). Analytical methods, intercalibration needs, and data interpretation have been considered by individual authors (Farrington et al., 1974), and groups (National Academy of Science, 1980). The approaches used in specific laboratories have been described (Bentz, 1976). The proceedings of the oil spill conferences sponsored by the American Petroleum Institute, the U.S. Environmental Protection Agency, and the U.S. Coast Guard constitute a useful collection of current research in oil pollution.

Because the pace of investigation has been intense and because the chronic and spill-related inputs of petroleum have been so frequent and often dramatic, a great deal has been learned about the impact of petroleum on the coastal environment during the decade:

Table 4. Concentrations of Fluoranthene and Pyrene in Mussels and Oysters (ppm-10^{-6} g/g Dry Weight)

	East and Gulf Coast, 1976-1977	
Station	Fluoranthene	Pyrene
Main		
Blue Hill Falls	.005	.003
Cape Hewagen	.084	.052
Massachusetts		
Cape Ann	.012	.066
Boston	.240	.329
Rhode Island		
Narragansett Bay	.026	.025
New York		
Manhasett Neck	.114	.381
Herod Point	.034	.021
Virginia		
Cape Charles	.047	.019
Lynnhaven Bay	.106	.062
North Carolina		
Hatteras Island	.042	.023
Beaufort	.169	.118
Georgia		
Sapelo Island	.005	.016
Savannah River	.192	.157
Washington		
Boundary Bay	3.35	1.54
Cape Flattery	0.34	0.45
California		
N. San Francisco	0.39	0.33
San Francisco	5.72	4.13
San Pedro Harbor	5.61	4.60
Louisiana		
Drum Bay	0.17	0.08
Texas		
Galveston	0.94	1.01
Lavaca Bay North	0.05	0.05

- Adequate information is on hand to design programs, large and small, to measure the impact of petroleum on coastal environments.
- There is an awareness that the total activities of the petroleum sector impact the coastal environment.
- Analytical chemistry techniques and expertise are at a high level so that chemical programs can be assured of reasonable success, although a need remains for intercalibration and standard samples.
- Concepts and techniques have been demonstrated for monitoring programs. Levels of petroleum in many of the coastal areas of the United States are fairly well known.
- Biological research has shown the general picture of petroleum toxicity; future studies may be expected to demonstrate the mechanisms for specific toxic action of a specific molecule type on a specific organism and to model the impact of petroleum on various ecosystems including benthic and pelagic ones.

- Promising starts have been made on research that measures the impact of petroleum on whole ecosystems or functional parts of ecosystems.
- Protocol for state and federal responses to major oil spills are beginning to take on a workable form.
- Management guidelines for the safe production, transport, and processing of petroleum in the coastal environment can be derived from the scientific base that has been established.

Given this optimistic status of knowledge about petroleum impacts, it is likely that future research and monitoring programs will provide an excellent basis for understanding how the flow of petroleum carbon relates to the flow of photosynthetic carbon in coastal ecosystems and what is means for marine life.

REFERENCES

Allen, H. 1971. Effects of petroleum fractions on the early development of a sea urchin. Mar. Pollut. Bull. 2:138–140.

Anderson, J.W. 1977. Effects of petroleum hydrocarbons on the growth of marine organisms. In: Petroleum hydrocarbons in the marine environment, A.D. McIntyre and K. Whittle, eds. Rapp. P.-v Reun., Cons. Int. Explor. Mer. 171:157–165.

Anderson, J.W., J.M. Neff, B.A. Cox, A.E. Tatem, and G.M. Hightower. 1974. Characteristics of dispersion and water-soluble extracts of crude and refined oils and their toxicity to estuarine crustaceans and fish. Marine Biology 27:75–88.

Arnold, C.R., S. Rabalais, and N. Wohlschlag. 1979. Effects of Mexican crude oil on redfish *(Sciaenops ocellata)* eggs and larvae. In: A study of the chemical composition and toxicity of petroleum from a Mexican spill. Final report to U.S. Coast Guard. University of Texas.

Atema, J., and L.S. Stein. 1974. Effects of crude oil on the feeding behavior of the lobster *Homarus americanus*. Environmental Pollution 6:77.

Bentz, A.P. 1976. Oil spill identification. Analytical Chemistry 48(6):454A–472A.

Berman, M.S., and D.R. Heinle. 1980. Modification of the feeding behavior of marine copepods by sublethal concentrations of water-accommodated fuel oil. Marine Biology 56:59–64.

Blumer, M., G. Souza, and J. Sass. 1970. Hydrocarbon pollution of edible shellfish by an oil spill. Marine Biology, International Journal on Life in Oceans and Coastal Waters 5(3):195–202.

Butler, J.N., B.F. Morris, and J. Sass. 1973. Pelagic tar from Bermuda and the Sargasso Sea. Bermuda Biological Station for Research, Special Publication No. 10.

Cerniglia, C.E., D.T. Gibson, and C. Van Baalen. 1980. Oxidation of naphthalene by cyanobacteria and microalgea. J. General Microbiology 116:495–500.

Clement, L.E., M.S. Stekoll, and D.G. Shaw. 1980. Accumulation, fractionation and release of oil by the intertidal clam *Macoma balthica*. Marine Biology 57:41–50.

Conomos, T.J. 1975. Movement of spilled oil as predicted by estuarine nontidal drift. Am. Soc. Limnol. Oceanogr. 20:159–173.

Conover, R.J. 1971. Some relations between zooplankton and Bunker C oil in Chedabucto Bay following the wreck of the tanker *Arrow*. J. Fish. Res. Board Canada 1327–1330.

Cucci, T.L., and C.E. Epifanio. 1979. Long-term effects of water-soluble fractions of Kuwait crude oil on the larval and juvenile development of the mud crab *Eurypanopeus depressus*. Marine Biology 55:215–220.

Davis, S.J., and C.F. Gibbs. 1975. The effect of weathering on a crude oil residue exposed at sea. Water Res. 9:275–285.

Dixit, D., and J.W. Anderson. 1977. Distribution of naphthalenes within exposed *Fundulus similus* and correlations with stress behavior. Proceedings from 1977 Oil Spill Conference: Prevention, Behavior, Control, Cleanup. American Petroleum Institute, U.S. Environmental Protection Agency, and U.S. Coast Guard, March 1977. pp. 633–636.

Donahue, W.H., R.T. Wang, M. Welch, and J.A.C. Nicol. 1977. Effects of water-soluble components of petroleum oils and aromatic hydrocarbons on barnacle larvae. Environ. Pollut. 13:187–202.

Eganhouse, R.P., and J.A. Calder. 1976. The solubility of medium molecular weight aromatic hydrocarbons and the effects of hydrocarbon co-solutes and salinity. Geochim. Cosmochim. Acta. 40:555–561.

Farrington, J.W., and J.G. Quinn. 1973. Petroleum hydrocarbons in Narragansett Bay. Estuarine and Coastal Mar. Sci. 1:71–79.

Farrington, J.W., and B.W. Tripp. 1977. Hydrocarbons in western North Atlantic surface sediments. Geochim. Cosmochim. Acta 41:1627–1641.

Farrington, J.W., J.M. Teal, J.G. Quinn, P.L. Parker, J.K. Winters, T.L. Wade, and K. Burns. 1974. Analyses of hydrocarbons in marine organisms: results of IDOE intercalibration exercises. National Bureau of Standards Publ. #409. Proceedings of a Symposium sponsored by the Intergovernmental Oceanographic Commission, the World Meteorological Organization, and the U.S. Department of Commerce, May 1974. pp. 163–166.

Fay, J.A. 1969. The spread of oil slicks on a calm sea. In: Oil on the sea. Plenum Press, NY. pp. 53–63.

Forrester, W.D. 1971. Distribution of suspended oil particles following the grounding of the tanker *Arrow*. J. Marine. Res. 29:151–170.

Fossato, V.U., and W.J. Canzonier. 1976. Hydrocarbon uptake and loss by the mussel *Mytilus edulis*. Marine Biology 36:243–250.

Frankenfeld, J.W. 1973. Weathering of oil at sea. Final report for Department of Transportation, U.S. Coast Guard, Washington, DC.

Gearing, J.N., P.J. Gearing, T. Wade, J.G. Quinn, and H.B. McCarty. 1979. The rates of transport and fates of petroleum hydrocarbons in a controlled marine ecosystem, and a note on analytical variability. Proceedings of the 1979 Oil Spill Conference, American Petroleum Institute, Washington, DC. pp. 555–564.

Gearing, P.J., J.N. Gearing, T.F. Lytle, and J.S. Lytle. 1976. Hydrocarbons in 60 northeast Gulf of Mexico shelf sediments: a preliminary survey.Geochim. Cosmochim. Acta. 40:1005-1017.

Goldberg, E.D. 1972. Marine pollution monitoring: strategies for a national program. Proceedings from a workshop, Santa Catalina Marine Biological Laboratory. National Oceanic and Atmospheric Administration. October 1972.

Goldberg, E.D., V.T. Bowen, J.W. Farrington, G. Harvey, J.T. Martin, P.L. Parker, R.W. Risebrough, W. Robertson, E. Schneider , and E. Gamble. 1978. The mussel watch. Environmental Conservation 5(2): 101-125.

Grassle, J.F., J.P. Grassle, L.S. Brown-Leger, N.J. Copley, and J.G. Smith Derby. 1980. Quantitative studies on benthic communities in experimental ecosystems. In: The fate and effects of chronic low level pollutants in marine ecosystems. Report from the Marine Ecosystems Research Laboratory. pp. 144–168.

Hargrave, B.T., and C.P. Newcombe. 1973. Crawling and respiration as indices of sublethal effects of oil and a dispersant on an intertidal snail *Littorina littorea*. J. of the Fish. Res. Board Canada 30:1789.

Harrison, W., M.A. Winnik, P.T.Y. Kwong, and D. Mackay. 1975. Crude oil spills—Disappearance of aromatic and aliphatic components from small sea-surface slicks. Environmental Science and Technology 9(3):231–234.

Horowitz, A., D. Gutnick, and E. Rosenberg. 1975. Sequential growth of bacteria on crude oil. Applied Microbiology 30:10–19.

Hunt, J.M. 1979. Petroleum geochemistry and geology. W.H. Freeman and Company, San Francisco.

International Decade of Ocean Exploration. 1972. Baseline studies of pollutants in the marine environment and research recommendations: the IDOE baseline conference, May 24–26, 1972, New York.

Jacobson, S.M., and D.B. Boylan. 1973. Effect of seawater soluble fraction of kerosene on chemotaxis in a marine snail *Nassarius obsoletus*. Nature 241:213.

Jobson, A., F.D. Cook, and D.W.S. Westlake. 1972. Microbial utilization of crude oil. Applied Microbiology 23:1082–1089.

Klug, M.J., and A.J. Markovetz. 1967. Degradation of hydrocarbons by members of the genus *Candida*. J. Bacteriol. 93:1847–1852.

Larson, R.A., D.W. Blankenship, and L.L. Hunt. 1976. Toxic hydroperoxides: photochemical formation from petroleum constituents. In: Sources, effects and sinks of hydrocarbons in the aquatic environment. Proceedings of a Symposium, American Institute of Biological Sciences, American University, Washington, DC., August 9–11, 1976. pp. 298–308.

Lee, R.F., R. Sauerhever, and A.A. Benson. 1972. Petroleum hydrocarbons: uptake and discharge by the marine mussel *Mytilus edulis*. Science 177:344–346.

Linden, O. 1976. Effects of oil on the amphipod *Gammarus oceanicus*. Environ. Pollut. 10:239–250.

Lowe, J.I., P.R. Parrish, J.M. Patrick, and J. Forester. 1972. Effects of the polychlorinated biphenyl Aroclor® 1254 on the American oyster *Crassostrea virginia*. Marine Biology 17:209–214.

Lytle, J.S. 1975. Fate and effects of crude oil on an estuarine pond. Proceedings from a conference on Prevention and Control of Oil Pollution. American Petroleum Institute, U.S. Environmental Protection Agency, and U.S. Coast Guard.

Mackay, G.D.M., A.V. McLeon, O.J. Betancourt, and D.B. Johnson. 1973. The formation of water-in-oil emulsions subsequent to an oil spill. Journal of the Institute of Petroleum 59:164–172.

MacLeod, W.D., D.W. Brown, R.G. Jenkins, L.S. Ramos, and V.D. Henry. 1976. A pilot study on the design of a petroleum hydrocarbon baseline investigation for northern Puget Sound and the Strait of Juan De Fuca. National Oceanic and Atmospheric Administration, Technical Memorandum ERL MESE–8.

McAuliffe, C.D. 1966. Solubility in water of paraffin, cycloparaffin, olefin, acetylene, cycloolefin, and aromatic hydrocarbons. J. Phys. Chem. 70:1267–1275.

McAuliffe, C.D. 1976a. Evaporation and solution of C_2 to C_{10} hydrocarbons from crude oils on the sea surface. Symposium on Fate and Effects of Petroleum Hydrocarbons in Marine Ecosystems and Organisms, November 10–12, 1976, Seattle, WA. pp. 363–372.

McAuliffe, C.D. 1976b. Surveillance of the marine environment for hydrocarbons. Marine Science Communications 2(1):13–42.

Mecklenburg, T.A., D.R. Stanley, and J.F. Karinen. 1977. Molting and survival of king crab *(Paralithodes camtschatica)* and coonstripe shrimp *(Pandalus hypsinotus)* larvae exposed to Cook Inlet crude oil water-soluble fraction. In: Fate and effects of petroleum hydrocarbons in marine organisms and ecosystems, D.A. Wolfe, ed. Pergamon Press. pp. 221–228.

Menzel, D.W., and J. Case. 1977. Concept and design: controlled ecosystem pollution experiment. Bull. of Mar. Sci. 27(1):1–7.

National Academy of Science. 1971. Marine environmental quality. Washington, DC.

National Academy of Science. 1975. Petroleum in the marine environment. Washington, DC.

National Academy of Science. 1980. The international mussel watch. Report of a Workshop sponsored by Environmental Studies Board Commission on Natural Resources, National Research Council. Washington, DC.

Neff, J.M., J.W. Anderson, B.A. Cox, R.B. Laughlin, S.S. Rossi, and H.E. Tatem. 1976. Effects of petroleum on survival, respiration and growth of marine animals. In: Sources, effects and sinks of hydrocarbons in the aquatic environment. Proceedings of a Symposium, American Institute of Biological Sciences, American University, Washington, DC., August 9–11, 1976. pp. 515–540.

Nicol, J.A.C., and R.T. Wang. 1977. Effects of fuel oil on sea catfish: feeding activity and cardiac responses. Bull. of Environ. Contam. and Toxic. 18(2):170–176.

Nicol, J.A.C., W.H. Donahue, R.T. Wang, and J.K. Winters. 1977. Chemical composition and effects of water extracts of petroleum on eggs of the sand dollar *Melitta quinquiesperforata*. Marine Biology 40:309–316.

Parker, C.M., M. Freegarde, and C.G. Hatchard. 1971. The effect of some chemical and biological factors on the degradation of crude oil at sea. In: Water pollution in oil, P. Hepple, ed. Institute of Petroleum, London. pp. 237–244.

Parker, P.L. 1974. Environmental design for an environmental program: hydrocarbon analysis in an oil producing area. Proceedings of Marine Environmental Implications of Offshore Drilling Eastern Gulf of Mexico, State University System of Florida, Inst. of Oceanography, St. Petersburg, FL. pp. 279–293.

Parker, P.L., R.S. Scalan, and J.K. Winters. 1976. Heavy hydrocarbons project-water zooplankton, neuston and sediment. In: Environmental assessment of the south Texas outer continental shelf chemical and biological survey component. Rept. to the Bureau of Land Management. pp. 443–530.

Parker, P.L., J.W. Winters, and J. Morgan. 1972. A baseline study of petroleum in the Gulf of Mexico. In: Baseline studies of pollutants in the marine environment (heavy metals, halogenated hydrocarbons and petroleum). The IDOE Baseline Conference, May 24–26, 1972, Brookhaven National Laboratory. pp. 555–583.

Pilson, M.E.Q., C.A. Oviatt, G.A. Vargo, and S.L. Vargo. 1979. Replicability of MERL microorganisms: initial observations. Advances in Marine Environmental Research. Proceedings of a Symposium, Narragansett, RI., September, 1979.

Roesijadi, G., D.L. Woodruff, and J.W. Anderson. 1978. Bioavailability of naphthalenes from marine sediments artificially contaminated with Prudhoe Bay crude oil. Environ. Pollut. 15:223–229.

Rossi, S.S. 1977. Bioavailability of petroleum hydrocarbons from water, sediments and detritus to the marine annelid *Neanthes arenaceodentata*. Proceedings of the 1977 Oil Spill Conference: Prevention, Behavior, Control, Cleanup. American Petroleum Institute, U.S. Environmental Protection Agency, and U.S. Coast Guard, March 1977. pp. 621–632.

Rossi, S.S., and J.W. Anderson. 1976. Toxicity of water-soluble fractions of No. 2 fuel oil and south Louisiana crude oil to selected stages in the life history of the polychaete *Neanthes arenaceodentata*. Bull. of Environ. Contam. and Toxic. 16:18.

Rossi, S.S., and J.W. Anderson. 1977. Accumulation and release of fuel-oil-derived diaromatic hydrocarbons by the polychaete *Neanthes arenaceodentata*. Marine Biology 39:51–55.

Rossi, S.S., J.W. Anderson, and G.S. Ward. 1976. Toxicity of water-soluble fractions of four test oils for the polychaetous annelids *Neanthes arenaceodentata* and *Capitalle capitata*. Environ. Pollut. 10:9-18.

Scott, B.F., and D.B. Shindler. 1978. Impact of crude oil on planktonic freshwater ecosystems. In: Proceedings of the Conference on Assessment of Ecological Impacts of Oil Spills. American Institute of Biological Sciences, Keystone, CO, June 1978.

Vandermeulen, J.H., and C.W. Ross. 1977. Assessment of cleanup tests of an oiled salt marsh —the *Golden Robin* spill in Miguasha, Quebec. Part I. Residual Bunker C hydrocarbon concentrations and compositions. Environmental Impact and Assessment Report. Fisheries and Environment Canada.

Walker, J.D., and R.R. Colwell. 1976. Enumeration of petroleum-degrading microorganisms. Appl. Environ. Microbiol. 31:198–207.

Walker, J.D., L. Cofone, Jr., and J.J. Cooney. 1973. Microbial petroleum degradation: the role of *Cladosporium resinae*. In: Proceedings of 1973 Joint Conference on Prevention and Control of Oil Spills. American Petroleum Institute, Washington, DC. pp. 821–825.

Walker, J.D., L. Petrakis, and R.R. Colwell. 1976. Comparison of the biodegradability of crude and fuel oils. Canadian Journal of Microbiology 22:598–602.

Wells, P.G. 1972. Influence of Venezuelan crude oil on lobster larvae. Mar. Pollut. Bull. 3:105–106.

Westlake, D.W.S., A. Jobson, R. Phillippe, and F.D. Cook. 1974. Biodegradability and crude oil composition. Canadian Journal of Microbiology 20:915–928.

Winters, J.K., and P.L. Parker. 1977. Water soluble components of crude oils, fuel oils, and used crankcase oils. Proceedings from 1977 Oil Spill Conference: Prevention, Behavior, Control, Cleanup. American Petroleum Institute, U.S. Environmental Protection Agency, and the U.S. Coast Guard, March 1977. pp. 579–582.

Winters, J.K., J.C. Batterton, and C. Van Baalen. 1977. Phenalen-l-one: occurrence in a fuel oil and toxicity to microalgae. Environ. Sci. and Tech. 11:270–272.

Winters, J.K., R. O'Donnell, J.C. Batterton, and C. Van Baalen. 1976. Water-soluble components of four fuel oils: chemical characterization and effects on growth of microalgae. Marine Biology 36:269–276.

Zobell, C.E. 1969. Microbial modification of crude oil in the sea. In: Proceedings of 1969 Joint Conference on Prevention and Control of Oil Spills. American Petroleum Institute, Washington, DC. pp. 317–326.

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